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Liner Location Risk and Cost Analysis Model

Pope-Reid Associates, Inc., St. Paul, MN

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AND COST ANALYSIS MODEL
DRAFT REPORT

SUBMITTED TO THE
OFFICE OF SOLID WASTE
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U.S. ENVIRONMENTAL PROTECTION AGENCY

(Under Contract No. 68-01-6621)
EPA Project Officer
Cliff Rothenstein

JANUARY 1985

PREFACE

This volume contains the Executive Summary and the text of the Liner Location Risk and Cost Analysis Model Draft Report, which was developed by a team of contractors under the direction of the U.S. Environmental Protection Agency, Office of Solid Waste, Economic Analysis Branch. The Report describes the model's framework, its applications and results of the first phase of development. A more detailed discussion of the the specific components of the model and the underlying assumptions is offered in the Appendix. Phase II of the model development is currently underway and will result in a final report which will be circulated for peer review.

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EXECUTIVE SUMMARY

The Liner Location Risk and Cost Analysis Model estimates the chronic risk to human health from land disposal of hazardous wastes under different technology, location, and waste stream scenarios. It also estimates the cost of different designs and sizes of landfills and surface impoundments. Together, this cost and risk information enables EPA to investigate the cost/risk and cost-effectiveness implications of various technology, waste, and location standards.

This report documents the first phase of the model's development. It describes the basic model and three policy applications of the model. The EPA project team is currently revising several components of the model and expects to make additional revisions in response to reviews of this report. The team will also perform additional policy analyses. Thus, the conclusions presented here are preliminary and exploratory. Furthermore, while the report and its appendices provide a sound fundamental explanation of the methodology and structure of the model, they are not entirely up to date on all points, since the model is currently being improved. EPA expects to prepare a second report explaining improvements in the model and the results of additional applications.

This summary provides a general description of the model, its structure, inputs, and outputs; a brief discussion of the three applications; and our plans for the future of this project.

1. THE LINER LOCATION RISK AND COST ANALYSIS MODEL COUPLES A RISK AND A COST MODEL

The Liner Location Risk and Cost Analysis Model links a risk and a cost model. The risk model simulates the chronic risk to human health from land disposal facilities (landfills and surface impoundments) with different design technology, location, and waste stream combinations. It integrates a series of submodels and algorithms that trace constituent releases from landfills and surface impoundments to their movement through the air and ground, then to resulting human exposures, and finally to the resultant human risk. To do so, model components predict:

- Releases to groundwater and subsurface transport. A failure and release submodel estimates facility failure and the quantity of leachate released; an unsaturated zone algorithm calculates the time required for the leachate to reach an underlying aquifer; and a saturated zone submodel calculates the time and concentrations of constituents reaching a down-gradient well.

- Releases to air and atmospheric transport. A volatilization algorithm calculates the quantity of constituents that volatilize over time; an atmospheric transport algorithm calculates the concentrations of these constituents at the exposure points.

- Human exposure. An exposure algorithm calculates the exposure from drinking water and from breathing constituents in air.

- Health risks. A hazard estimation submodel calculates expected cancer and noncancer risks from the exposures.

The model calculates risk over a 400-year time horizon. In doing so, it embodies many assumptions: facilities operate for 20 years with a 30-year post-closure period; contamination goes undetected and uncorrected; multiple contaminants do not interact; and few constituents degrade over time, with degradation beginning only after facility failure. The model also assumes that aquifers are homogeneous and isotropic. In general, we believe that the model overestimates risk.

The companion cost model estimates the costs of land disposal facilities with differing technologies and sizes. The results of the risk model combined with cost data from the cost model provide EPA with the capability to perform cost/risk and cost-effectiveness analysis.

1.1 Risk Model

The failure and release submodel estimates the probability and time of failure and the quantity of subsequent leachate release through the use of a Monte Carlo simulation model. The model uses a "fault tree" structure; that is, it traces each possible failure event (leakage from the facility) from all possible combinations of more elementary events which could combine to cause failure: infiltration of liquid, liner failure, etc. The model assumes that each of these elementary events occurs at random, following specified probability distributions. The failure probabilities over time are the average of probabilities for a large number of facilities. Using this fault tree analysis, the model predicts the year of failure and the release rates for ten different landfill and surface impoundment designs in five different climates. Liner and cover type and material define facility designs, while average net infiltration of moisture per year and depth to ground water correspond to climate type.

In the release component of the submodel, we assume that dissolution in leachate releases waste constituents to ground water and that volatilization releases constituents to air. Although the failure submodel is stochastic and provides a probability distribution for liner failure, the "downstream" exposure modeling components are all deterministic, and thus produce risk estimates for a given release rate starting at a specific time in the simulation period. To link the failure component of the model with the subsequent exposure submodel, we used the convention that release to the unsaturated zone begins to occur when the probability of failure exceeds 0.01. Preliminary sensitivity analysis shows that over the 400-year simulation period, risks are not very sensitive to the probability of failure selected to trigger releases.

The submodel performs a yearly mass balance to simulate the release of constituents. The annual potential release mass to groundwater is equal to the leachate volume multiplied by the aqueous solubility. Potential release mass to air is calculated using empirical formulas for volatilization rate. After failure, release of constituent occurs until all of the mass has been leached or volatilized from the facility. Although the failure and release submodel predicts different probabilities of failure and average release volumes for different designs, the submodel does not remove a corresponding amount of constituent mass from facilities with intact leachate collection and removal systems. Because single-lined land disposal facilities generally fail sooner than double-lined facilities, the submodel may understate the contribution of delaying liner failure to reducing risk.

The exposure modeling for the groundwater route examines both the unsaturated zone and the saturated zones. It uses the McWhorter-Nelson wetting front equation to calculate the delay from the time liners fail to the time that the leachate comes in contact with an underlying aquifer. The mass breaking through the unsaturated zone then disperses through and dilutes in groundwater. Using an adaptation of Prickett's Random-Walk Particle Tracking Model, we model groundwater concentrations over a 400-year time horizon at distances corresponding to locations of wells. We calculate these concentrations for nine different groundwater flow scenarios that involve variations in aquifer configuration and velocity; all scenarios apply only to homogeneous, isotropic aquifers.

The rate of flow of the groundwater and retardation caused by adsorption to organic carbon in the earth materials govern the transport of constituents through the unsaturated and saturated zones. Retardation calculations use the Freundlich isotherm algorithm, which assumes direct proportionality between constituent velocity (relative to the groundwater front) and the water: organic carbon partition coefficient.

To estimate the transport of volatile constituents, we used the OTS Atmospheric Transport Model (ATM-SECPOP). We calculated annual average ground level concentrations at ten sample scenarios that represent meteorological conditions at locations in the U.S. to determine whether site-specific differences in meteorology result in significant differences in estimated concentrations. They did not; the site-to-site variation in concentrations was small and concentration was linearly proportional to emissions rate. We thus calculated a linear scaling factor to translate volatilization rate into concentrations at various distances from the source, and then used these factors to estimate annual average ambient air concentrations for populations exposed to emissions of volatile constituents.

The groundwater concentrations cause exposure doses as a result of the assumption that all receptors drink two liters of water per day. Air concentrations cause exposure doses as a result of the assumption that receptors have an air intake of 22 cubic meters per day. In both cases we assume the average weight of receptors is 70 kg, and in most cases we assume equal absorption through all exposure routes. The lifetime dose is the running 70-year average dose.

To produce a risk estimate, we multiply dose by a dose-response factor. The hazard submodel uses the same risk estimation formula as the OSW Risk/Cost Analysis Model. The Liner Location Risk and Cost Analysis Model uses continuous functions to model potency for these threshold pollutants; these functions predict less risk below no-effect levels than more traditional methods such as acceptable daily intakes (ADIs) would predict. Thus, while we treat risk from carcinogenic contaminants in a manner consistent with traditional EPA methods, our approach to contaminants which pose threshold risks differs from the traditional approach.

The hazard submodel reports risk estimates as average risk (i.e., the time-weighted lifetime risk over the 400-year period) and maximum risk (i.e., the risk for the 70-year period with the highest running average exposure). By using population data, the submodel can also calculate the additional number of cases of adverse health effects over the 400-year simulation period.

Finally, a single computer model links all the algorithms and submodels described above.

1.2 Cost Model

To calculate costs for landfills and surface impoundments we:

- Specified designs and sizes;
- Estimated the engineering costs for those designs and sizes; and
- Discounted the engineering costs to produce comparable economic measures.

We used EPA's Facilities Design Computer Cost Model to estimate construction and closure costs for landfills and surface impoundments of different liner configurations and varying sizes. In implementing the model, we estimated initial capital costs (land, constructing the first year's cell for landfills or excavating and installing liners for surface impoundments, building the infrastructure, etc.), annual capital costs (building the next year's cell and closing the last year's cell for landfills), closure costs (covering the final cell for landfills or covering the surface impoundment, decontaminating equipment, monitoring, and certifying closure), and O&M costs associated with the technical aspects of operating the facility. For surface impoundments, we also estimated the frequency and cost of dredging. Since our costs were developed to compare relative technology costs, we did not include O&M costs for recordkeeping, reporting, financial responsibility, or any other costs that do not change as the stringency of liner regulations changes. Thus, for a particular liner design, our costs underestimate the total cost of disposing of hazardous waste.

Important assumptions underlie these engineering cost estimates. First, we assumed that facilities are able to use on-site, recompacted clay in all designs which require clay. If they are unable to do so, engineering costs will be from 12 to 53 percent greater than our calculations indicate, depending on facility design and size. We assume that all surface impoundments and smaller landfills are on-site and will not require fences and gates, office buildings, scales, and equipment that the larger off-site facilities will require.

To convert engineering costs to economic measures we calculated total present values for each facility design and size by discounting the engineering costs in each year of the facility's life and summing the discounted costs over all years. We used a three percent real discount rate to calculate present values.

OSW's standard assumptions for economic analysis pertaining to facility life and operating procedures applied to the cost model: surface impoundments and landfills both operate for 20 years; operators open the first cell in a landfill at the beginning of year one and cover it at the beginning of year two, and cover the last cell at the beginning of year 21; and operators line surface impoundments at the beginning of year one and cover them at the beginning of year 21.

After calculating the present values, we also annualized them. Costs are often unevenly distributed through time; annualizing spreads them evenly over the life of the facility. We annualized by multiplying the present value of each facility size and design by a capital recovery factor. With a three percent real rate of return and a 20-year facility life, the capital recovery factor is 0.0672.

Finally, we calculated the annualized present value on a per unit basis by dividing the annualized cost for each facility size and design by its capacity. This results in annualized costs per ton for landfills, and annualized costs per acre and per gallon for surface impoundments.

2. APPLICATIONS FOR THE MODEL EVALUATE THE COST/RISK IMPLICATIONS OF TECHNOLOGY STANDARDS AND COMPARE RISKS OF DIFFERENT ENVIRONMENTAL SETTINGS AND DIFFERENT WASTES

We have applied the Liner Location Risk and Cost Analysis Model on several occasions, both to test and debug it and to use it for policy analysis. Given the complexity of the model and restrictive time schedules, we often performed the policy analysis closely following and sometimes concurrent with testing and debugging the model. Also, those instances where we know the distributions around input values, we have not carried this information through to final estimates but have only determined point estimates. In other instances, we do not currently have estimates of the distribution of values, and have not performed sensitivity analyses to determine how changing values would change our risk estimates. Therefore, we have not yet calculated error bands or confidence intervals for the risk and cost estimates. We therefore view the results of the first three sets of applications as tentative and exploratory rather than conclusive.

In the first application, we evaluated national technology standards from a cost/risk perspective; in the second we compared the risk of environmental settings, holding technology and waste streams constant; and in the third, we compared the risk of different waste streams, holding technology and environmental setting constant. In general, we found that risks of land disposal vary widely depending on wastes, environments, and technologies. We also found that risks are more sensitive to wastes and environments than they are to technology.

2.1 Evaluating National Technology Standards

In this application of the Liner Location Risk and Cost Analysis Model, we used two different sets of input data. The first set represents a cross section of 17 landfills and 31 surface impoundments using actual data for the facilities wherever possible. While we think these sites comprise a reasonably representative sample of landfills and surface impoundments, they are not a statistically valid random sample.

Among the "actual sites," only two of the landfills and five of the surface impoundments have nearby populations using ground water as their source of drinking water. Of these, only one landfill and three impoundments show any risk, even for the unlined design. Moreover, more stringent liner designs (single, double, and triple liners) provided little incremental risk abatement at these facilities. Consequently, requiring liners at all of these facilities would result in a very high cost per case avoided.

Because these estimated risks are largely driven by the number of receptors, we repeated the analysis using 44 additional landfill sites and assuming that receptors exist at all sites. (Figures showing these results are labeled "Augmented Sites.")

Among the sites augmented with hypothetical populations, risks are substantially larger. Almost two-thirds of the unlined facilities which were modeled showed some level of risk, although most of this risk was concentrated at a small number of sites and slightly more than one-third of these unlined facilities had no risk. Further, the results indicate that liner technology is not particularly effective in abating these risks at all modeled sites. Even so, combining technology cost information with the risk information suggests that the cost/risk ratios of requiring liners at all modeled sites is favorable. However, tailoring the requirements to site-specific factors that increase risk rather than requiring the same standards for all facilities could provide even better cost/risk ratios.

2.2 Comparing the Risk of Environmental Settings

In this application we examined the effect that environmental setting has on risk. Holding technology constant (double synthetic liners) we examined the distribution of risk to exposed populations under varying climate, hydrogeologic setting, and well distance scenarios. Thus, for three waste constituents (chromium, carbon tetrachloride, and parathion) we examined risk to individuals under 135 possible combinations of environmental setting (five climates, nine hydrogeologic settings, and three well distances). We performed the simulations for landfills and for surface impoundments.

The results indicate that differences in environmental settings lead to large differences in risks to exposed individuals from disposing of a waste stream with a given management practice and technology. Approximately one-third of the scenarios for chromium in surface impoundments produce risks of 10^{-1} (one in ten) to exposed individuals while about one-third of the scenarios produce no risks. For landfills, almost 40 percent of the scenarios produce risk of 10^{-1}, but about 25 percent of the scenarios produce no risk at all.

The results for carbon tetrachloride show a similar pattern of both high and low risk. However, the overall risks are lower (nearly three-quarters of the scenarios produced no risk) because carbon tetrachloride is less mobile than chromium and exposures occur in fewer scenarios.

None of the parathion scenarios result in any exposures; due to its low mobility and high rate of degregation, no significant concentration breaks through the unsaturated zone during the simulation period.

In addition to showing that environmental setting causes large differences in risks, this application also begins to suggest how the three environmental factors we varied are important in determining risk. First, drier climates produce less risk because there is less leachate and therefore releases after failure are smaller. Next, the results illustrate the effect of distance on risk as zero risk scenarios are far more likely at longer distances from wells. Finally, in comparing scenarios with slow-moving aquifers to scenarios with faster ones, we observe that for persistent constituents in the fast aquifers, the peak and average constituent concentrations are lower, the period of exposure is shorter, and the average risk ranges from two to as much as five orders of magnitude lower. For degradable constituents, the period of exposure is shorter in fast aquifers but the peak and average concentrations may be higher and consequently, risks may also be higher.

In summary, this application shows that risks are extremely sensitive to environmental setting. Juxtaposed with the evaluation of national technology standards, it shows risk is far more dependent on environmental setting than on liner technology.

2.3 Comparing the Risk of Different Wastes

To compare the risks of different wastes, we simulated risks for 15 waste streams in landfills and 30 waste streams in surface impoundments, holding design, liner technology, and environmental setting constant. We selected waste streams by examining responses to OSW's mail survey. We modeled the risk for a waste stream either in landfills, surface impoundments, or both if (1) it appeared in landfills, surface impoundments, waste piles, injection wells, or land treatment as one of any facility's ten largest waste streams; (2) it can be managed in landfills or surface impoundments; and (3) we could characterize its constituents and find the physicochemical properties necessary to simulate its risks.

Our results showed that there is wide variability in risk for different waste streams holding all other factors constant. In landfills, eight of 15 wastes we modeled showed average risks of less than 10^{-9} (one in a billion), yet all but one of the remaining seven waste streams showed risks of 10^{-4} (one in ten thousand) or greater. In surface impoundments, nine of 30 waste streams showed risks of less than 10^{-9}, while 17 showed risks of 10^{-4} or more.

This application juxtaposed with the evaluation of national technology standards indicates that risk is far more sensitive to changes in waste stream composition than to changes in technology.

3. PLANS FOR THE PROJECT CALL FOR IMPROVING THE MODEL, VALIDATING IT, AND ISSUING A PHASE II REPORT

We have not currently completed all the sensitivity analyses, nor have we examined all the assumptions and limits to determine the constraints they impose on future use of the model for policy purposes. As the tasks are completed, EPA will issue a second report.

3.1 Improving and Validating the Model

In building and applying the model we have noticed several areas that we can improve; basic areas of effort include the failure and release submodel, the subsurface transport submodel, the exposure submodel, the hazard model, and the cost model.

In the failure and release submodel, we will (1) perform sensitivity analyses to determine how important the underlying probability distributions are in determining failures and releases; (2) modify the assumption that leachate collection systems remove water, but not hazardous waste from facilities; and (3) change the basic structure of the submodel so we can report statistics associated with the estimates of failure percentages and leachate volume releases.

We are also working on ways to model constituent degradation prior to release from the site as well as other modifications to the estimates of waste mass available to enter the unsaturated zone.

The subsurface transport submodel will be expanded to consider attenuation in the unsaturated zone, dispersion into a third dimension, dilution of contaminants at the well site, and transport through fractured, not homogeneous and isotropic, aquifers.

We will also relax our worst-case assumptions about exposure: first, that people drink contaminated water, even if contaminants exceed their odor, taste, and physical appearance thresholds; and second, that groundwater monitoring is not in place.

In the hazard submodel we will further evaluate some of the basic assumptions underlying the risk estimation algorithm for noncarcinogens, compare our risk estimates with those derived by other methods, and determine the sensitivity of results to changes in input values.

In the cost model, we will include the cost of treating leachate from the facility, whether on or off-site, the cost and effectiveness of taking corrective or remedial actions as well as the cost of repairing or replacing covers and other facility components.

Because of the difficulty in measuring some of the complex processes we model, it is not possible to verify directly the performance of the model, but we can infer the accuracy of several key elements by reviewing some types of empirical data. As one example, we plan to review results of ongoing air monitoring at landfills and surface impoundments in order to verify that our approximations of air emissions are realistic. Some of the other material we will review includes data on effectiveness of groundwater monitoring and contaminant concentration in groundwater at CERCLA sites.

3.2 Phase II Report

When we complete the work described above, EPA will issue and circulate for peer review the final Liner Location Risk and Cost Analysis Model Report. This will present the model, modified to incorporate the improvements discussed previously and comments from this first round of review.

1. INTRODUCTION

The Liner Location Risk and Cost Analysis Model contains two linked models. One simulates the individual and population risks that land disposal of hazardous wastes poses under different combinations of wastes, environmental settings, and technologies; the other estimates the costs of land disposal for different technology and facility size combinations. Combining these two models allows the Environmental Protection Agency (EPA) to calculate cost/risk and cost-effectiveness estimates. This chapter provides a brief history of the project and an overview of the rest of the report.

1.1 History of the Project

On July 26, 1982, EPA promulgated interim final regulations for new and existing land disposal facilities. Because the Agency was under court order (State of Illinois v. Gorsuch) to promulgate these regulations for existing facilities, it was exempt from the section of Executive Order 12291 which requires agencies to prepare a regulatory impact analysis (RIA) to accompany major regulations. When the Agency promulgated the regulations, it had not investigated, formally, the costs and risks of alternative standards for liners and location standards. There were many unanswered questions: Are single synthetic liners more effective than clay liners? If so, are they worth the additional cost? Are double synthetic liners significantly more protective of the environment than single liners and are they worth their additional cost? Are there areas of the country or waste streams for which liners are not needed to protect the environment, or for which the risks are too high even with liners in place?

To answer questions like these, EPA committed to study the costs and risks of alternative liners for different wastes and in differing environmental settings. The Agency stated in the preamble to the regulations that "EPA hopes to improve these regulations in the future by developing . . . specific standards that are tailored to specific wastes, locations, and management practices." (47 FR 143 at 32278) The Office of Solid Waste (OSW) initiated the Liner/Location Project in part to fulfill this commitment, and in part to support an RIA for the specific tailored standards that it hoped to develop later.

The Economic Analysis Branch (EAB) within the Waste Management and Economics Division is responsible for this project. EPA retained Pope-Reid Associates, Inc. (PRA), for the failure and release analysis and for the engineering cost analysis; Geraghty & Miller, Inc. (G&M), for subsurface transport modeling; ICF Inc., for characterizing wastes and constituents, exposure modeling, integrating the various submodels into a computerized risk model, and--together with Environ Corp.--effects modeling; and Sobotka and Co., Inc. (SCI), for assisting EPA in the overall project coordination and integration, and for the economic and cost/risk analyses. Although each firm was assigned areas of prime responsibility, the firms worked cooperatively and all participated in work outside their area of prime responsibility.

EAB assembled the project team early in 1983. By December of 1983, the linked models had become operational, and the process of testing, debugging, and performing applications began. The EAB project team is still in the process of doing all three, as well as improving the model, and expects to continue for some time.

1.2 Overview of the Report

The remainder of this report is comprised of four chapters and seven appendices. Following is a brief description of each of the remaining chapters.

Chapter 2: THE RISK MODEL presents the methodology of each component of the risk model, together with the inputs, results, and limitations. It discusses the failure and release submodel, subsurface transport modeling, atmospheric transport modeling, exposure modeling, and effects modeling, as well as the computer structure linking the pieces of the risk model.

Chapter 3: THE COST MODEL AND COST/RISK ANALYSIS discusses the methodology of the cost model, and its inputs, results, and limitations. In addition, this chapter provides a discussion of the cost/risk modeling.

Chapter 4: APPLICATIONS OF THE MODEL contains the results of three separate applications of the model. We applied the model to (1) determine from a cost and risk perspective the policy implications of alternative liner requirements for land disposal facilities, (2) compare the risks of land disposal in various environmental settings, holding wastes and technology constant, and (3) compare the risks of disposing of different wastes, holding technology and environmental setting constant.

Chapter 5: PLANNED MODEL IMPROVEMENTS AND APPLICATIONS describes where the project is headed. The first section of this chapter discusses improvements we are planning to components of the model as well as improvements actually in progress. Then it discusses model validation, additional applications, and plans for the next report from this project.

Volume II of this report is comprised of seven appendices which are as follows:

A - Failure and Release Submodel	Pope-Reid Associates, Inc.
B - Subsurface Transport Submodel	Geraghty & Miller, Inc.
C - Exposure Submodel	ICF Inc.
D - Hazard Submodel	ICF Inc.; Environ Corporation
E - Cost Model	Pope-Reid Associates, Inc.; Sobotka & Co., Inc.
F - Waste and Constituent Characterization	ICF Inc.
G - Structure and Function of the Model	ICF Inc.

2. THE RISK MODEL

This chapter discusses the risk portion of the Liner Location Risk and Cost Analysis Model, which estimates chronic risk to human health over a 400-year simulation period. The risk model is comprised of several component parts, each of which requires specific inputs and embodies particular assumptions.

The first component is a *failure and release submodel* that predicts when land disposal facilities fail and the quantities of leachate they release afterwards. A subsurface transport model then simulates the movement of the chemicals released into the ground: first an *unsaturated zone algorithm* calculates how long it takes the leachate to travel through the unsaturated material underlying the facility and then a *saturated zone submodel* predicts contaminant concentrations in groundwater at selected distances and times once contaminants reach the groundwater.

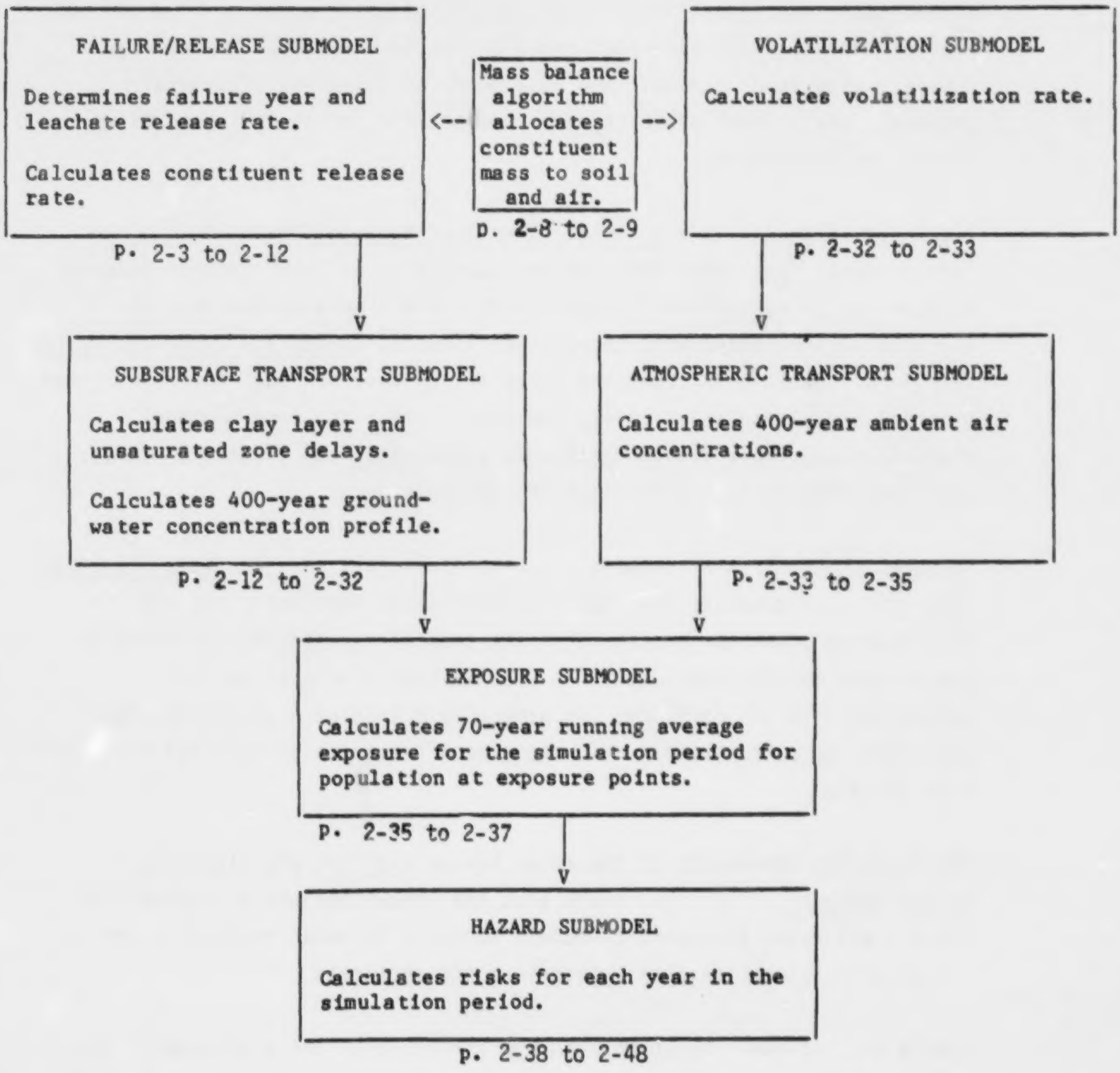
To model the release of chemicals to the atmosphere we use a *volatilization algorithm* that predicts the mass of constituents that enter the air. A mass balance algorithm ensures that the mass of constituents released to the air and unsaturated zone is no greater than the total mass of constituents in the facility. An *atmospheric transport algorithm* then estimates the concentrations of volatilized chemicals at selected distances from the site.

The final two components of the model are an *exposure algorithm* and a *hazard submodel*. The first translates the concentrations estimated from the air and water transport submodels to doses to human receptors, and the second calculates risk from exposure to the chemicals.

Figure 2-1 outlines the components and structure of the risk model. The inputs required to drive these component submodels and algorithms include

Figure 2-1

STRUCTURE OF THE RISK MODEL



FAILURE/RELEASE SUBMODEL

Determines failure year and leachate release rate.

Calculates constituent release rate.

p. 2-3 to 2-12

Mass balance algorithm allocates constituent mass to soil and air.

p. 2-8 to 2-9

VOLATILIZATION SUBMODEL

Calculates volatilization rate.

p. 2-32 to 2-33

SUBSURFACE TRANSPORT SUBMODEL

Calculates clay layer and unsaturated zone delays.

Calculates 400-year groundwater concentration profile.

p. 2-12 to 2-32

ATMOSPHERIC TRANSPORT SUBMODEL

Calculates 400-year ambient air concentrations.

p. 2-33 to 2-35

EXPOSURE SUBMODEL

Calculates 70-year running average exposure for the simulation period for population at exposure points.

p. 2-35 to 2-37

HAZARD SUBMODEL

Calculates risks for each year in the simulation period.

p. 2-38 to 2-48

such disparate information as physicochemical characteristics of the constituent chemicals, their toxicities, the climate near the facility, and the design and size of the facility. We discuss the inputs and intermediate outputs for each model component in later sections of this chapter, as noted in Figure 2-1.

This analysis incorporates many model-specific assumptions that we address in detail later in this chapter. However, some of the most important ones are that facilities operate for 20 years with a 30 year post-closure period, contamination goes undetected and uncorrected and there is no pre-existing contamination, multiple contaminants do not interact, all adverse health effects have equal weight, the entire quantity of waste deposited into the facility is available to leave the facility by volatilizing or by leaching after facility failure, and aquifers are homogeneous and isotropic. In general, we expect that the model is conservative and overestimates risks.

This chapter provides a brief description of the methodology of the components of the risk model, their inputs and assumptions, and the results of applying the methodology. Appendices A through D, F, and G contain more complete descriptions.

2.1 Failure Analysis - Release to Ground Water

The objectives of the failure analysis were to determine the probability of various types of failures (release of leachate from the facility) at selected times for different types and designs of facilities in different climates, and to estimate the quantity of leachate released. Appendix A: Failure Analysis provides a more detailed discussion.

2.1.1 Inputs

The failure analysis covered four types of facilities: hazardous waste landfills, surface impoundments, waste piles, and land treatment facilities. For applications, we have focused on only two of these four--landfills and surface impoundments. Various combinations of liner and cover type and material defined ten different landfill and ten different surface impoundment designs. Average annual net infiltration of moisture and depth to groundwater both directly correspond to climate type; the five categories ranged from "arid" (no infiltration, deep unsaturated zone) to "humid" (20 or more inches of infiltration per year, shallow unsaturated zone). The model user specifies facility design, facility size, and climate type as basic inputs.

The failure and release submodel also requires inputs that characterize the waste streams present in a given disposal facility, including a listing of which toxic agents (constituents) comprise the waste streams. Appendix F describes in detail the specific methods used to characterize wastes for this project. The basic model inputs include the identity of each constituent present in the facility, disposal rates, and aqueous solubilities of each constituent.

In this submodel, solubility determines the mass of each constituent that land disposal facilities release following a liner failure. Mass release rate is the product of flow passing through the liner and the aqueous solubility of the constituent--that is, the submodel assumes that leachate is saturated with the constituent. Several standard secondary references contain the solubilities of organic constituents. The solubilities of inorganics are generally much more variable, depending on pH, ligands present, oxidation/reduction potential, and other factors. However, the model assumes a constant solubility value for each inorganic constituent.

2.1.2 Fault Tree Analysis

The failure analysis submodel utilizes a "fault tree" structure, tracing possible failure events (release of leachate from the facility) from all possible combinations of more elementary events--infiltration of liquid, liner failure, etc.--which could combine to cause failure. At each of these nodes in the tree structure, the submodel calculates or estimates a probability of failure at seven relevant "benchmark" times (after 5, 10, 20, 35, 50, 100, and 200 years).

The fault tree methodology is appropriate for estimating the probability of any single overall failure event, since it takes into account the probabilities of all possible combinations of the more elementary events which could lead to the failure event of interest. It is important to remember, however, that certain elementary events, such as flooding, could cause more than one type of failure event or occur more than once; the submodel underestimates the probability of occurrence of these types of events.

Probability Distributions. The failure submodel uses probability distributions for two kinds of factors: whether an event occurs, and if it does, the volume of liquid entering or exiting the facility.

The submodel uses the fault tree structure to compute the probabilities of failure events which occur if some sequence of prerequisite elementary events have occurred. Each of the elementary events is assumed to occur at random according to one of three types of probability distributions: "triangular," "binomial," and "time-dependent binomial."

A triangular distribution uses a specified time interval, T, and assigns a linearly increasing probability of occurrence to the event of interest as we move from the initial time to T. The event is certain to occur by T and is far more likely to occur as time approaches T: for example, the probability that the event occurs within the first one-fourth of the time interval T is 0.0625, while the probability of occurrence in the last one-fourth of the time interval is 0.4375. We used this distribution for most events involving components which tend to wear out, such as covers and liners, where T is the estimated maximum lifetime of the component.

A binomial distribution describes an event which has the same probability (p, for example) of occurring in any time interval (year, in this case). This distribution describes location-specific events such as whether facilities are located in areas prone to earthquakes and floods. It also describes time-independent events such as operator error.

A "time-dependent binomial" is a truncated binomial distribution; the binomial distribution, as before, determines whether an event occurs within T years. Then, however, the model randomly selects the year in which it occurs from years 1 through T. This distribution differs from the usual binomial in that the event can only occur once within the T-year time interval.

We developed volumetric distributions for all elementary events that are associated with a volume of liquid (e.g., volume of run-on during a flood, annual infiltration after cap failure, proportion of free liquids in wastes, groundwater table fluctuations) based on engineering judgment or, in a few instances, actual field data on liquid volumes. We assumed a uniform distribution with lower and upper bounds in most cases. Once the event 'occurred', the model used the Monte Carlo model to simulate a volume of liquid in the relevant compartments by 'sampling' from the volume distribution. The Monte Carlo model is explained below.

Monte Carlo Simulation of Overall Outcome. Monte Carlo simulation is an appropriate technique for estimating the overall outcome of a sequence of events for which probabilities are known for each individual event. It consists of "sampling," using a computer-based random number generator, from the probability distributions already assumed or estimated for the elementary events. By repeating this process many times and observing the proportion of those repetitions in which certain events occur, we obtain an estimate of their relative frequency--that is, the probability of occurrence of that sequence of events. For specific points in time (benchmark times) the model also calculates releases of leachate to the unsaturated zone if failure occurs.

To estimate the probability of failure at each of the benchmark times for each land disposal technology, the modeling method follows three steps:

(1) It estimates the probabilities of elementary events--that is, those for which probability distributions have been specified using the three probability distributions described earlier.

- For events we assumed to have a "binomial" distribution, a random number generator "draws" a number between 0 and 1. If this random number is less than p, the (assumed) probability of occurrence, then the event occurs; otherwise it does not occur. This distribution is used for events such as whether the facility is in an area prone to earthquakes.

- For events we assumed to have a "time-dependent binomial" distribution, where p is the (assumed) probability of occurrence in any year, the probability of occurrence within T years (where T is the benchmark period) is calculated using the formula Prob (event within time T) = $1 - (1-p)^T$. The computer generates a random number, and the event occurs if the random number is less than the calculated probability of occurrence. If the event occurs, then the year of occurrence is estimated by drawing randomly from the years 1 through T, each having equal probability of selection. This distribution is used for such events as run-on flooding.

- For events described by the "triangular" distribution with parameter M (estimated lifetime), a random number between 0 and 1 is generated. The square root of this random number is the proportion of the interval from the start (time 0) to time M which elapses before the event occurs. (Note that events of this type are certain to occur exactly once in the interval from 0 to M, most likely near the end of the interval.) This distribution is used for events including the aging and wearout of liners (35-year lifetime assumed for synthetic liners), aging of leachate systems due to corrosion (75 years), damage to liner during inspection (37 years), earthquakes (100 years), and loss of cover due to erosion (1.5 to 40 years, depending on the type of cover).

(2) Next, it evaluates the remaining events in the fault tree. Because we have specified probability distributions for "elementary" events, the model can use them to determine whether each elementary event occurred or did not, and use that information to evaluate whether subsequent events occur. Some events can occur before year T only if two or more other events have all occurred ("AND gate," in fault tree terminology) before year T; others occur if at least one of the more elementary events has occurred ("OR gate"). As events whose occurrence depends upon whether other events have occurred are evaluated, still other events further up the fault tree can be evaluated in turn.

(3) Finally, it evaluates whether release occurs. At the top of the fault tree, the top event, "release," either occurs or does not occur. For those scenarios in which release does occur, mass balance analysis determines the volume released, as described in the next section.

These three steps were repeated 1000 times for each design in each climate at each benchmark time, the only variation being the random sampling from the specified probability distributions. Dividing the number of runs which resulted in releases by the total number of runs (1000) yields the estimate of the probability of release for the scenario in question.

2.1.3 Modeling of Mass Balance Given Release

For each run in which a release occurs in the fault tree simulation, the model estimates the volume of released leachate in four steps.

(1) The time from failure to the benchmark is calculated. For example, if the benchmark time is 100 years and the facility fails in year 60, the time period from failure to the benchmark time is 40 years.

(2) The potential infiltration volume is calculated using the annual net infiltration rate for the climatic region, the surface area of the facility, and the time from failure to the benchmark period.

(3) The events that lead to the release determine the total volume of water potentially entering the facility from all sources--runoff, flooding, free liquid wastes (in surface impoundments), infiltration, and groundwater table fluctuations. Reducing this amount by a proportion corresponding to the estimated efficiency of components which reduce the volume of liquid inputs (such as covers and run-on controls) yields the total volume of liquid in the facility.

(4) The total volume released is estimated by subtracting from the total volume of liquids in the facility, the quantity of liquid the leachate collection system removes, and, if applicable, the volume of leachate stored in the clay layer beneath the facility.

The average total leachate volume is calculated by dividing the total leachate volume for runs with releases by the number of runs. The annual average release rate is calculated by dividing average total leachate volume by the number of years in the benchmark period.

2.1.4 Reporting of Results

For each land disposal technology, design, climate, and benchmark period, the failure and release submodel reports three results:

(1) Probability of a failure event during the benchmark period,

(2) Average volume of leachate released, during the benchmark period, and

(3) Average annual volume of leachate released during the benchmark period.

To convert the probability of failure into a deterministic outcome for the later models (transport, exposure, etc.), we assume that release begins when the probability of failure exceeds a pre-selected trigger probability, usually 0.01. Preliminary sensitivity analyses using alternative trigger points indicate that risk estimates for most variable combinations are probably not very sensitive to the probability of failure selected as the trigger point; they vary by a factor of 2 to 3.

2.1.5 Results

The failure and release modeling results show that the age of the facility, climate (net infiltration), and the facility design all affect failure and release rates. Over the 200 year simulation period, most facilities fail. Those facilities located in wetter climates fail earlier than those in drier climates and release more leachate once they fail; facilities with more preventive technology last longer and release less leachate once they fail.

The time elapsed clearly affects the failure rate as many of the events which can lead to failure are time-dependent. Accordingly, failure rates increase with time (see Appendix A). Although release rates generally follow the same pattern, releases occasionally decrease over time when catastrophic events occur early in the simulation causing large initial releases, followed by less severe, gradual leakage. Finally, after the post-closure care period ends (assumed to be year 50), the failure percentages and release rates in landfills and surface impoundments increase markedly. Some of this is due to the long time period encompassed by the next benchmark period (50 years as opposed to 5, 10, or 15 previously) but there is also some effect resulting from neglect.

In general, both failure rates and release rates increase with net infiltration. With no net infiltration, nearly every landfill remains intact over the simulation period. Surface impoundments have a higher failure rate than landfills, most noticeably over the first twenty years because liquids are present in the impoundment during this period. All facilities' failure rates increase as infiltration increases; even a small volume of infiltration will result in a lower level of performance. Release rates are also very sensitive to net infiltration indicating that the integrity of the cover may play a crucial role in governing release volumes.

Facilities with more preventive technology have lower failure and release rates. For example, double liners are superior to single ones in preventing failure and release and higher failure rates occur in facilities without drainage collection systems, leachate collection systems, or liners. For landfills and for surface impoundments, single clay liners appear to be about as effective as single synthetic liners in terms of life expectancy and in preventing release. In landfills, double clay liners are slightly more effective than double synthetic liners, while in surface impoundments double clay and double synthetic liners appear to perform similarly. Given the uncertainty in the analysis, the differences between clay and synthetic may not be meaningful. Also, these results probably depend on the assumptions underlying the probability distributions for clay and synthetic liners.

2.1.6 Assumptions and Limitations

The results of this submodel depend on certain critical assumptions.

- The engineering judgments used to define designs, specify probabilities of various events, and describe combinations of events which would lead to failure are clearly very important, and different judgments in these areas could affect the results. In particular, the probability distributions and expected lifetimes specified for various events could affect the overall conclusions to a significant extent.

- Releases to groundwater are governed by the solubility of the constituents and the flux of leachate. This is clearly inaccurate for constituents that are present as free liquids. For inorganics, use of a single solubility value masks the wide variability in potential leaching rates. This approach also ignores solvent-waste interactions, which could mobilize the low-solubility, low-mobility constituents.

- All of the waste mass disposed of in surface impoundments is retained in the impoundment. Except in arid climates, virtually all lined surface impoundments discharge to surface water. The effluent is likely to contain some of the waste mass that was introduced with the influent, and therefore the mass the impoundment retains would actually be less than the total mass input. Moreover, we assume that when the impoundment closes after 20 years of operation, the wastes are stabilized. Subsequently, leachate through the stabilized waste mass can solubilize any wastes not leached during the impoundment's operating life.

In summary, all the "downstream" risk model components are affected by the failure submodel. The failure analysis is based on a number of important assumptions and limitations, in both methodology and supporting data. We are in the process of evaluating some of these assumptions and limitations (as described in Chapter 5) and will soon be able to appraise more fully the expected accuracy and utility of the failure submodel.

2.2 Subsurface Transport Modeling

The subsurface transport submodel of the risk model describes the mass transport of contaminants through both the unsaturated and the saturated zones. It traces contaminants from their mass loading into the unsaturated zone, through the saturated zone, and ultimately predicts concentrations over time in wells down gradient from a hypothetical land disposal facility. The submodel's results can be scaled to describe subsurface transport at real facilities. The following sections summarize both the unsaturated zone submodel and the saturated zone submodel. More detailed discussion appears in Appendix B: Subsurface Transport Modeling.

Modeling subsurface transport in both the unsaturated and the saturated zones requires information about the persistence and mobility of contaminants' constituent chemicals. Persistence in groundwater is a function of the rates of hydrolysis, biodegradation, and other processes. Rather than estimating persistence at individual sites, the submodel uses input values. These assume that the dominant transformation processes are first-order half-life values for constituents in the W-E-T [1] data base.

Mobility of a constituent in groundwater is generally controlled by the strength of sorption reactions between the constituent and the earth materials. The stronger the sorptive force, the slower the constituent moves with respect to the velocity of the groundwater field. We calculated retardation factors (Rd's) for various constituents, using a Freundlich isotherm approach that describes the relative velocity of the constituent compared to the groundwater front. The mobility of a specific contaminant in a subsurface system can be described by its velocity with respect to the velocity of water according to the equation:

$$Rd = V_W/V_c$$

where Rd is the retardation factor, V_W is the velocity of water and V_c is the velocity of the contaminant.

2.2.1 Unsaturated Zone Modeling

The unsaturated zone is the soil layer above the water table in which the pores in the soil are only partially filled with water. We selected the McWhorter-Nelson wetting front model to simulate travel time from the base of the facility to the water table. Unlike the saturated zone submodel, this model does not simulate the process of dispersion.

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[1] RCRA Risk-Cost Analysis Model, Phase III Report. March, 1984. ICF Incorporated. This model is commonly referred to as the Waste-Environment-Technology (W-E-T) model.

The model simulates only homogeneous, isotropic hydrogeologic scenarios in the unsaturated zone. In addition, all of the unsaturated zone scenarios contain groundwater flow only in the vertical direction. We also assumed that the earth material in the unsaturated zone is the same as that in the top layer of the saturated zone.

Contaminant transport is represented by the movement of a wetting front generated by the release of leachate from a land disposal facility. The travel time for the wetting front through the unsaturated zone is related to the difference in water content between the layers immediately above and below the front, the distance traveled in the unsaturated zone, the leakage rate (Darcian flux) at the facility boundary, and the retardation of contaminant flow by soil adsorption according to the equation:

$$t_r = [L*(\Theta_f - \Theta_r)/q] * Rd$$

where:

t_r = Contaminant travel time from the facility liner to the top of the saturated zone;
Θ_f = Water content above the wetting front;
Θ_r = Water content below the wetting front;
L = Distance from the liner to the top of the saturated zone;
q = Leakage rate, or Darcian flux, at the liner;
Rd = Retardation factor.

The first expression, $L*(\Theta_f-\Theta_r)/q$, defines the transport time of water over the distance L. Θ_r is the volumetric soil moisture capacity, and is dependent on total porosity, permeability, and pore size. Θ_f depends on Θ_r and the rate of leakage into the unsaturated zone, q. The rate of leakage is based on the failure and release submodel and is held essentially constant between benchmark years for the unsaturated zone modeling. More precise modeling would reflect the fact that q varies considerably depending on daily infiltration rates: our current treatment therefore introduces some error in our calculation. L is dependent on the thickness of the unsaturated zone and the depth of the disposal facility. Typical values for thickness, soil moisture, porosity, permeability, and soil pore size used in the model appear in Figure 2-2. These values are based on published data, data obtained from site-visit surveys for this project, and our professional judgment.

The transport time for water is then adjusted to reflect the transport time for a specific contaminant by multiplying it by the contaminant's retardation factor, Rd. This adjustment is necessary to account for the effect of adsorption to soil particles on the mobility of contaminants in both the unsaturated and saturated zones.

A contaminant's retardation factor depends on contaminant characteristics, such as hydrophobicity and polarity, and soil characteristics including fraction of organic carbon, porosity, and bulk mass density. To estimate constituent retardation factors we assumed an average earth material with constant values for these soil parameters. These values are based on accepted average values for a variety of material types. This resulted in contaminant Rd values ranging from 1 to over 10,000. In the unsaturated zone we used the Rd's calculated for each constituent.

Unsaturated Zone Model Results. After adjusting wetting front travel times to reflect contaminant mobility, we found that the McWhorter-Nelson Model results depend on seven factors:

- Thickness of the unsaturated zone,
- Retardation coefficient of the contaminant,
- Specific discharge of leachate from the contaminant source,
- Porosity of the unsaturated zone,
- Soil moisture content in the unsaturated zone,
- Unsaturated zone permeability, and
- Pore size distribution.

Figure 2-2

RANGE OF MODEL PARAMETER VALUES FOR UNSATURATED ZONE MODEL

Parameter	Aquifers[a]	Nonaquifers[b]
Thickness (ft)	1 - 500[c]	1 - 500[c]
Soil Moisture (%)	0.1 - 0.2	0.05 - 0.45
Total Porosity (%)	0.1 - 0.4	0.05 - 0.50
Permeability (cm/sec)	10^{-1} - 10^{-4}	10^{-4} - 10^{-7}
Pore-Size Index	2 - 5	1 - 2

[a] Aquifers are defined as having flow velocities of at least 1 m/yr.

[b] Nonaquifers have flow velocities less than 1 m/yr.

[c] Site-specific thickness (depth to water table) as shown in Table 4, Appendix B.

The first three factors have the greatest effect on contaminant transport time. Transport time is directly proportional to the thickness of the unsaturated zone and the retardation coefficient and inversely proportional to the specific discharge of leachate from the contaminant source. The remaining four parameters have minor effects on travel time, and the averages of the values on Figure 2-2 are used as inputs into the model for these parameters.

Unsaturated Zone Model Limitations. To model contaminant transport in the unsaturated zone using the wetting front model it was necessary to make several simplifying assumptions.

- No contaminant interactions occur. Contaminant interaction, for example the mobilization of hydrophobic organics by organic solvents, can reduce contaminant transport time.

- A Freundlich isotherm approach describes transport. This approach is applicable when sorption reactions are fast and reversible, and when the solute is present at dilute concentrations. It is not clear that these conditions apply.

- No dispersion occurs. Ignoring it can result in an overestimate of travel time.

2.2.2 Saturated Zone Modeling

The model also tracks the transport of contaminant mass in the saturated zone, and predicts contaminant concentration over time at distances down gradient of land disposal facilities. To do so, the model uses a set of generalized groundwater flow fields that represent a range of groundwater velocities and aquifer configurations (e.g., single layer, double layer). The model then estimates the concentrations at down gradient wells (60 m, 600 m, and 1500 m from the source) for a unit loading rate (1 kg/yr), for constituents having four different retardation factors, in each of the nine generalized flow fields. It estimates concentrations over a 200-year time horizon for each of the 108 well distance - mobility class - flow field combinations (3 x 4 x 9). These concentration profiles are stored as a reference file in the overall risk model, and concentrations are scaled up or down in proportion to the loading rate entering the saturated zone. Thus, the saturated zone model does not currently accept site-specific hydrogeologic parameter values. Instead, it uses the generic well distance - mobility class - flow field combinations that most closely resemble conditions at a site, and estimate risks using the generic conditions.

The following sections explain how we developed the generic groundwater flow fields, well distances, and mobility classes, and how we applied the Random-Walk Solute Transport Model developed by Prickett, Naymik, and Lonnquist to estimate concentrations in our generic environmental settings.

Developing Generic Groundwater Flow Fields and Well Distances. To model the mass transport of contaminants in the saturated zone over the range of diverse hydrogeologic conditions found in the United States, we developed nine representative generic groundwater flow fields. Developing these generic flow fields was a three step process. First, we performed a literature survey of groundwater supply reports to determine representative ranges for parameters describing aquifer configurations, aquifer sizes, and groundwater flows. Second, we used the aquifer configuration data ranges to construct eight generic hydrogeological settings. Third, we matched these eight settings with representative aquifer size and groundwater flow data to create our nine generic flow fields. (Some pairs from the eight settings have highly similar flows and were combined into one generic field.) These nine flow fields represent the dominant types of subsurface flow found in homogeneous, isotropic aquifers developed for water supply in parts of the United States where land disposal facilities are commonly located.

In step one, we selected a total of 67 groundwater supply reports which provide the data necessary for developing the generic scenarios. The locations from which we gathered these reports represent the range of hydrogeologic conditions found in the United States in eleven groundwater regions identified by the United States Geological Survey.

We grouped these data into two categories of hydrogeologic information:

(1) Aquifer configuration information: earth material present, type of aquifer systems present (i.e., aquifer or nonaquifer), and the sequence of aquifers and nonaquifers within the uppermost 300 feet of the saturated flow system.

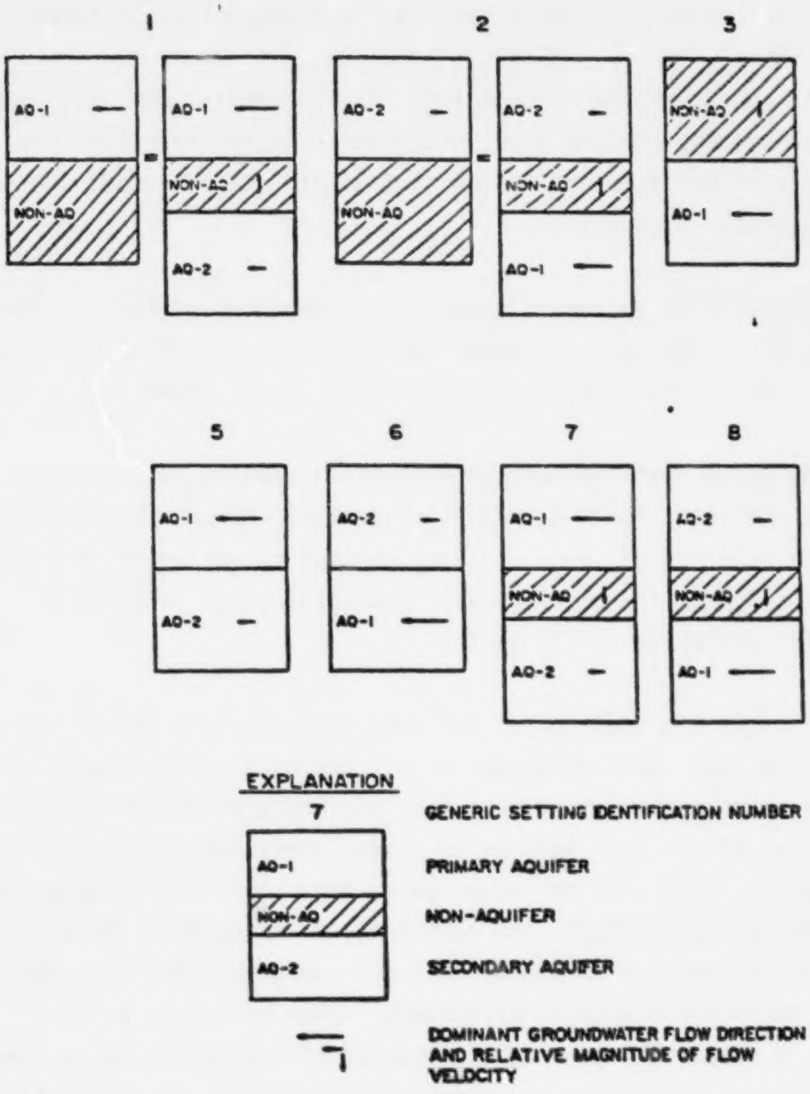
(2) Groundwater flow system information: dimensions of total system, system discharge mode, thickness of each main velocity layer, and horizontal and vertical flow velocities in each layer.

In step two, we used the aquifer configuration data to develop eight generic hydrogeologic settings based on the types of aquifer systems present, the layering of these systems, and the direction of groundwater flow in each layer. The eight generic hydrogeological settings characterized are shown in Figure 2-3.

Each of the eight hydrogeological settings is conceptualized as a two dimensional vertical plane passing through the center of a land disposal facility to a depth equal to the base of shallow groundwater circulation beneath the site. In each setting an unsaturated zone immediately beneath the disposal facility overlies a saturated zone containing groundwater flow. As in the unsaturated zone submodel, we assume that the subsurface materials in our modeled saturated zone are homogenous and isotropic with respect to their hydrogeologic parameters. (The exception to this assumption is dispersivity, which is modeled as homogenous but anisotropic in the saturated zone.) The layering sequence in each of the settings consists of one or more layers, with each layer being a primary aquifer,

Figure 2-3

EIGHT GENERIC HYDROGEOLOGIC SETTINGS SHOWING SEQUENCE OF WATER-BEARING MATERIALS AND DOMINANT GROUNDWATER FLOW DIRECTIONS


1
AQ-1
NON-AQ
=
AQ-1
NON-AQ
AQ-2
2
AQ-2
NON-AQ
=
AQ-2
NON-AQ
AQ-1
3
NON-AQ
AQ-1
4
NON-AQ
AQ-2
5
AQ-1
AQ-2
6
AQ-2
AQ-1
7
AQ-1
NON-AQ
AQ-2
8
AQ-2
NON-AQ
AQ-1
EXPLANATION
7
GENERIC SETTING IDENTIFICATION NUMBER
AQ-1
PRIMARY AQUIFER
NON-AQ
NON-AQUIFER
AQ-2
SECONDARY AQUIFER
DOMINANT GROUNDWATER FLOW DIRECTION AND RELATIVE MAGNITUDE OF FLOW VELOCITY

secondary aquifer, or a nonaquifer. We assume primary aquifers contain materials with higher permeabilities and groundwater flow velocities than secondary aquifers.

In step three, we used the groundwater flow system data collected in step one to convert these eight hydrosettings into our nine generic flow fields. We developed horizontal and vertical velocity distributions from these data, and then used these distributions together with professional judgment to assign appropriate velocity values to each layer. Thicknesses were also assigned to each layer in the generic hydrogeologic settings based on typical values found in the data base for each layer type.

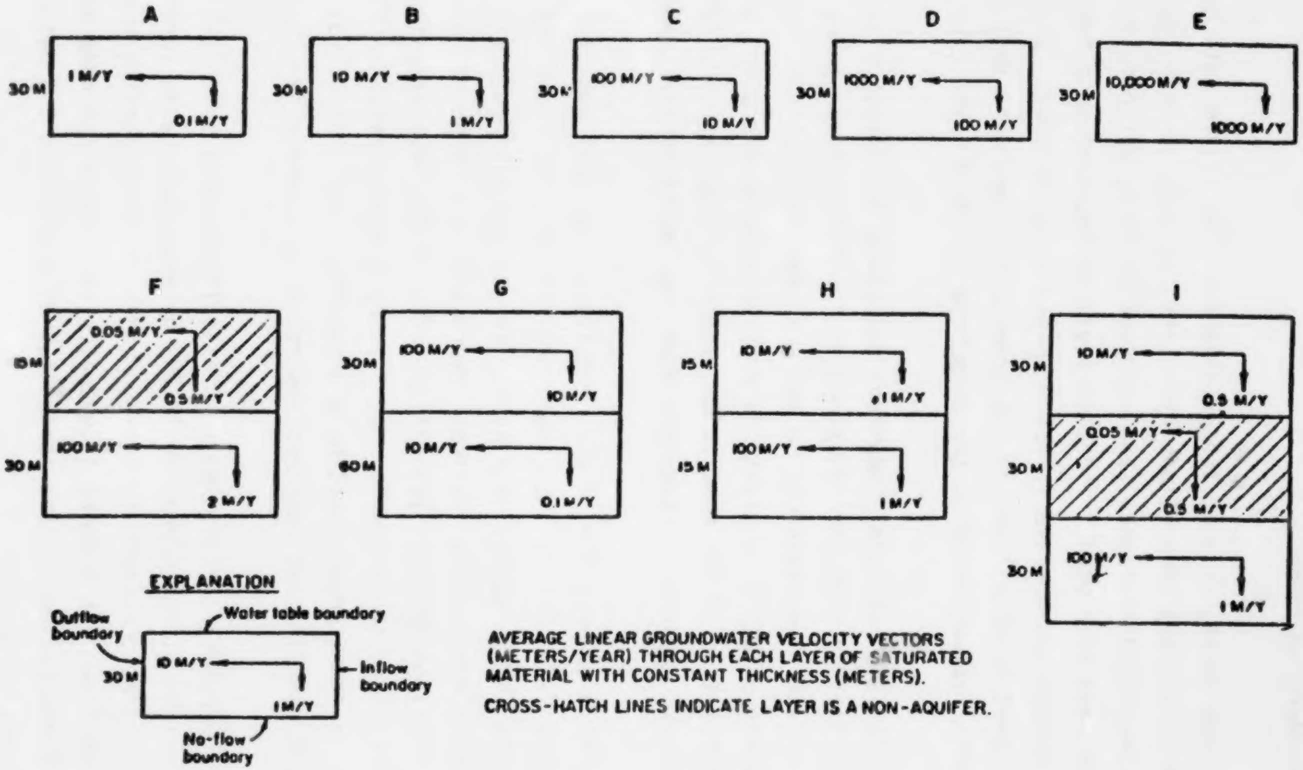
The final dimensional component necessary to characterize the groundwater flow fields, flow path distance to the point of exposure, was determined from reviewing distances to wells downgradient of existing facilities, the density of wells in areas where private wells are used, and the predictive capabilities of the model. Distances selected are 60, 600, and 1500 meters. Appendix C, Exposure Modeling, contains additional detail.

The nine generic flow fields resulting from this approach are displayed in Figure 2-4. Due to the common occurrence of a single shallow aquifer system in the survey data base, five of the nine velocity fields are single-layer aquifer systems, representing a range of horizontal and vertical groundwater velocities. One of the remaining four flow fields contains a single aquifer overlain by a nonaquifer, two flow fields contain two aquifers unseparated by a nonaquifer, and another multiple-layer flow field contains two aquifers separated by a nonaquifer layer.

The horizontal velocities in the aquifers range over five orders of magnitude (1 meter/year to 10,000 meters/year), with vertical aquifer velocities one-tenth to one-hundredth of corresponding horizontal velocities. We assumed groundwater velocities for nonaquifers to equal 0.05 meters/year in the horizontal direction and 0.5 meters/year in the vertical direction.

Figure 2-4

NINE GENERIC GROUNDWATER FLOW FIELDS TO BE USED IN TRANSPORT MODELING SCENARIOS


A
30 M
1 M/Y
0.1 M/Y
B
30 M
10 M/Y
1 M/Y
C
30 M
100 M/Y
10 M/Y
D
30 M
1000 M/Y
100 M/Y
E
30 M
10,000 M/Y
1000 M/Y
F
15 M
0.05 M/Y
0.5 M/Y
30 M
100 M/Y
2 M/Y
G
30 M
100 M/Y
10 M/Y
60 M
10 M/Y
0.1 M/Y
H
15 M
10 M/Y
.1 M/Y
15 M
100 M/Y
1 M/Y
I
30 M
10 M/Y
0.5 M/Y
30 M
0.05 M/Y
0.5 M/Y
30 M
100 M/Y
1 M/Y
EXPLANATION
Outflow boundary
Water table boundary
30 M
10 M/Y
1 M/Y
Inflow boundary
No-flow boundary

AVERAGE LINEAR GROUNDWATER VELOCITY VECTORS (METERS/YEAR) THROUGH EACH LAYER OF SATURATED MATERIAL WITH CONSTANT THICKNESS (METERS).

CROSS-HATCH LINES INDICATE LAYER IS A NON-AQUIFER.

Developing Generic Mobility Classes. As discussed in the unsaturated zone submodel section, the model measures mobility by calculating the retardation coefficient for a contaminant in a standard environment.

For modeling purposes we grouped the contaminants of concern into the following four broad classes based on their Rd value:

Class	Rd Range	Median Rd
1	≤ 10	1.3
2	10 and ≤ 100	32
3	>100 and ≤ 10,000	360
4	> 10,000	160,000

The model then used the median Rd value for each class as the value describing the mobility of contaminants in the modeled generic flow field scenarios. It generally ignores chemical interactions which could serve to increase mobility for all contaminants except those in class 4. Early model results indicated that mobility class 4 constituents would not move a significant distance (i.e., would not reach a well 60 m downgradient) during the standard 200 year saturated zone model scenario. In some runs, to simulate a scenario in which the mobility of these contaminants was increased due to chemical interaction, we have modeled class 4 materials as if they had class 3 mobility.

Contaminant Transport Modeling. We selected the Random-Walk Particle Transport Model to model concentrations in our generic contaminant transport scenarios. The Random-Walk Model is an analog groundwater model, which simulates the transport of dissolved pollutants as if they were particles. Particles move in two stages during each time step. In stage one, the "advective stage," each particle in a subregion moves in the direction of groundwater flow. The distance a particle moves is dependent on the velocity of the contaminant it is representing relative to groundwater (i.e., its mobility) and the length of the time step. In stage two, the "dispersive stage," each particle disperses randomly based on the values of exogenous dispersion coefficients (which are a function of mobility and dispersivity), and the particle velocity.

Dispersivity affects contaminant transport only in the saturated zone portion of the submodel. We based the dispersivity values chosen for both aquifer and nonaquifer layers on the general groundwater literature. For aquifer layers, we assumed that horizontal (longitudinal) dispersivity was primarily dependent on the length of the flow path. Dispersivities in this direction were assigned values of 2, 4 and 7.5 meters for the 60, 600, and 1500 meter flow paths respectively, reflecting a linear increase in dispersivity with an increase in flow path. We assigned vertical dispersivities in aquifer layers values equal to one-tenth of the horizontal dispersivity and assumed they were independent of aquifer thickness.

For the low-permeability nonaquifer layers we assumed dispersivity to be independent of flow path length and layer thickness. We assigned dispersivity values (irrespective of length of flow path) of 0.5 meters for dispersivity in the vertical direction and 0.1 meters in the horizontal direction.

Dispersion in the submodel is limited to two dimensions by assuming that dispersion in the transverse direction was negligible. Mobility affects dispersivity in both aquifer and nonaquifer layers. This effect is adjusted during the dispersive stage of contaminant transport in the submodel.

Overall, the dispersivities in this submodel are conservative, resulting in an increase in the relative contaminant concentrations found in the generic flow scenarios, and in the wells downgradient.

The concentration of a contaminant in the groundwater which the model predicts at a specified exposure well depends on the transport factors discussed above. In calculating this concentration we assume that a hypothetical well at the exposure point draws water from the entire thickness of an aquifer layer and that the contaminant mass is evenly distributed vertically at that point. The concentration at a particular time step is determined by counting the number of particles passing an exposure well distance point during the time step and converting this number to a mass flux at that point. This mass flux is then divided by the volumetric flux of water entering the exposure well during the same time step, yielding a concentration in mg/liter.

To calculate the mass loading of a surface water body from the aquifer of concern, the model assumed that particles reaching the water body were instantaneously released into it. The model counted the number of particles entering the water body during a time step and converted this number to a mass load.

Results. The Liner Location Risk and Cost Analysis Model does not incorporate the complete saturated zone submodel. Instead, it simulates a large number of combinations of input variables to produce summary outputs, or "look-up tables," which the submodel stores. It then selects the appropriate entries in the tables for each simulation, and scales them depending on the mass loading rate.

The nine generic flow fields, three exposure point distances, and four mobility classes describe 108 contaminant transport scenarios. In the subsurface transport submodel, 62 of the 108 scenarios result in contaminants initially arriving at the point of exposure or discharge after the 200 year simulation period for contaminant transport. We did not perform model simulation runs for these 62 scenarios. Of the remaining 46 scenarios for which model runs were completed, 28 represent single-layer systems and 18 are multilayer scenarios. A list of each of these scenarios together with their contaminant breakthrough times (the time of first exposure) at the exposure point and the peak breakthrough concentrations for a 1 kg input of mass within the first year appears in Figure 2-5. The

Figure 2-5

CONTAMINANT BREAKTHROUGH TIMES AND PEAK CONCENTRATIONS COMPUTED BY THE RANDOM-WALK MODEL

Scenario Identification Number*	Starting Time (yr)	Ending Time (yr)	Peak Conc. Time (yr)	Peak Conc. (mg/l)
A-1-60	26	> 200	71	2.4
B-1-60	4	20.5	9	2.4
B-2-60	61	> 200	201	0.12[a]
C-1-60	0.35	2.5	1.5	1.5
C-2-60	8	45	19	0.10
C-3-60	96	> 200	186	0.0085
D-1-60	0.06	1.12	0.34	0.20
D-2-60	0.80	5.2	2.4	0.09
D-3-60	9	51	21	0.0085
E-1-60	< 0.04	1.0	0.04	0.013
E-2-60	0.14	1.36	0.42	0.019
E-3-60	0.75	5.2	2.4	0.0089
B-1-600	61	106	81	0.60
C-1-600	5.5	12.5	8.5	0.57
C-2-600	131	> 200	191	0.026
D-1-600	0.5	2.0	1.4	0.15
D-2-600	13.5	29	20.5	0.028
D-3-600	151	> 200	201	0.0023[a]
E-1-600	0.06	1.06	0.78	0.017
E-2-600	1.2	3.5	2.4	0.015
E-3-600	16	81	22	0.0023
B-1-1500	141	> 200	196	0.30
C-1-1500	14	27	19.5	0.29
D-1-1500	1.3	3.5	2.1	0.15
D-2-1500	35	67	48.5	0.014
E-1-1500	0.14	1.2	0.52	0.020
E-2-1500	3.8	7.4	5.4	0.0087
E-3-1500	37	71	51	0.0010
F-1-60	16	135[b]	41	0.070
F-1-600	37	111[b]	49	0.096
F-1-1500	45	101[b]	59	0.095
G-1-60-Up	0.40	3.0	1.25	1.2
G-1-60-Lo	1.25	9.75[b]	1.5	0.11
G-1-600-Up	6	17	9	0.08
G-1-600-Lo	8	101	76	0.14

Figure 2-5 (Continued)

Scenario Identification Number	Starting Time (yr)	Ending Time (yr)	Peak Conc. Time (yr)	Peak Conc. (mg/l)
G-1-1500-Up	19[c]	> 200	151	0.0011
G-1-1500-Lo	23	> 200	186	0.11
G-2-60-Up	11	57	21	0.10
G-2-60-Lo	27[c]	> 200	143	0.013
G-2-600-Up	136	> 200	201	0.0053[a]
G-2-600-Lo	151[c]	> 200	191	0.0067
G-3-60-Up	121	> 200	196	0.0072
G-3-60-Lo	146[c]	> 200	191	0.0027
H-1-60-Up	3.25	19.75	7.75	4.1
H-1-60-Lo	< 1.75	16.75[b]	8.5	0.11
H-1-600-Up	23.5	46.5[b,c]	38.5	0.032
H-1-600-Lo	8.5	98.5[b]	23.5	0.13
H-1-1500-Up	26	56[b,c]	36	0.069
H-1-1500-Lo	21	136	36	0.090
H-2-60-Up	91	> 200	181	0.18
H-2-60-Lo	21	> 200	156	0.0037
I-1-60-Up	4.5	19[b]	9	2.4
I-1-60-Lo	0[d]	0	0	0
I-1-60A-Up	6	21	11	2.0
I-1-60A-Lo[e]	86	> 200	151	0.017
I-1-600-Up	51	111[b]	81	0.25
I-1-600-Lo	41	> 200	155	0.015
I-1-1500-Up	157	> 200	199	0.040
I-1-1500-Lo	51	> 200	163	0.016

* Scenario identification numbers are of the form: generic flow field - constituent mobility class - distance in meters to exposure point - aquifer layer (if the flow field has more than one layer: Up = Upper Aquifer Layer; Lo = Lower Aquifer Layer).

[a] Maximum concentration at end of simulation period represents peak breakthrough concentration.

[b] End of breakthrough defined as first occurrence of two consecutive time steps with zero concentrations. See Appendix C for explanation.

[c] Appearance of contaminants from another model layer attributed to dispersion in a direction against the hydraulic gradient vector.

[d] Contaminants enter the lower aquifer beyond a distance of 60 m.

[e] Groundwater flow direction in lower aquifer is reversed to cause breakthrough at 60 m exposure point.

starting time for contaminant breakthrough in these scenarios ranges from a fraction of one year to over 150 years. Breakthrough extends beyond the 200 year time frame in some instances, with the length of the exposure period ranging from as little as one year to over 180 years. Predicted peak concentrations in wells (per kg of waste released per year) range over several orders of magnitude, from 0.001 to 4.1 mg/liter.

We then use the well concentrations estimated for each time step by the Random-Walk Model to describe the subsurface transport of a contaminant in a particular generic scenario (i.e., groundwater flow field, mobility class, exposure distance) by constructing a standard breakthrough curve, as illustrated in Figure 2-6. These curves are plots of concentration at the point of exposure (breakthrough point) vs. simulation time. The shape and amplitude of the curves reflect the physical conditions being simulated and provide an estimate of the time of breakthrough, the duration of the exposure, and the maximum exposure concentration. Mass loading curves plot the cumulative mass loaded to a surface water body vs. time.

Several parameters in the model have major effects on the breakthrough time, peak contaminant concentration, and duration of breakthrough:

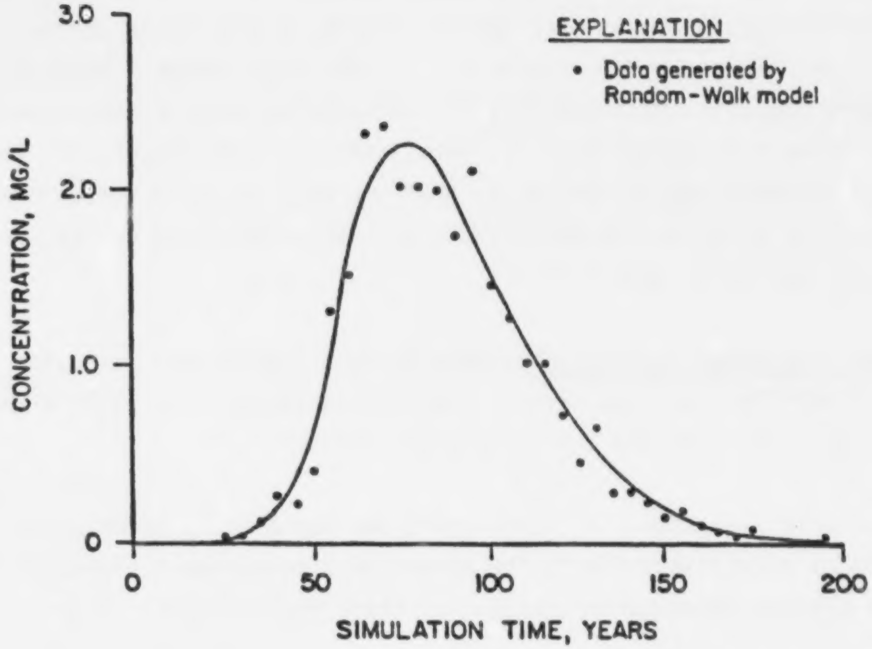
- Aquifer thickness: Peak concentrations of contaminants are inversely proportional to the thickness of the aquifer layer.

- Travel distance to breakthrough point: The peak concentration at an exposure point decreases as the distance to that point increases. Also, the duration of the exposure increases as travel distance increases.

- Groundwater velocity: As the velocity of groundwater increases, contaminants reach the exposure points earlier, exposure periods are shortened, and concentrations decrease.

Figure 2-6

STANDARD BREAKTHROUGH CURVE FOR TRANSPORT
MODEL SCENARIO A-1-60



- Contaminant retardation factor: Increasing contaminant retardation factors result in decreased peak concentration of contaminants, and an increased breakthrough period.

- Low velocity layers above aquifers of interest: For these scenarios there is an increase in the time to exposure, a lengthening of the exposure period, and a decrease in peak concentrations.

The authors of the Random-Walk code recently modified it to account for the partitioning of contaminant mass between the solid and liquid phases in an aquifer. Our current model runs do not include this change. However, preliminary analysis indicates that the adjustments made in contaminant velocity using retardation factors, serve indirectly to account for the effects of chemical partitioning, so that our results yield contaminant concentrations which are reasonably close to those produced by the current version of the Random-Walk code.

Saturated Zone Model Limitations. Developing and using the saturated zone transport submodel required several simplifying assumptions which affect the submodel's results and limit its applicability.

Using conservatively small dispersivities and assuming no transverse dispersivity effectively limits the spread of a contaminant plume and thus probably overestimates concentrations at the exposure wells.

Using a single mobility factor (Rd) to describe the mobility of a group of contaminants decreases the accuracy of the estimated concentrations by biasing the resulting contaminant breakthrough times, peak concentrations, and exposure length towards those contaminants which have Rd's near the middle of their class. Assuming that there are no contaminant mobilization interactions probably understates contaminant transport times and peak concentrations.

In addition to these assumptions concerning the physical/chemical aspects of the groundwater flow scenarios, we made some important assumptions regarding the subsurface transport system used in the model. We assumed that:

- Wells draw from only the contaminated aquifers of interest. This ignores water well dilution which would occur if the wells were to draw from more than one layer in a groundwater system.
- The contaminant mass in an aquifer is evenly distributed vertically, and wells draw from the full thickness of the aquifer. Contaminants may not distribute in this manner, particularly over short travel distances. Also, many wells are not likely to be screened over the full thickness of an aquifer.
- Degradation occurs only for contaminants whose decay rates are readily available. Many organic chemicals decay at rates that are so slow that they are difficult to measure, but which could dramatically reduce concentrations over a 200 year time frame.
- Drawdown effects do not occur. The cone of depression resulting from drawdown could result in a flux of uncontaminated water into the aquifer of concern, leading to lower contaminant concentrations.
- The exposure points are located directly downgradient from the contaminant source. This maximizes contaminant concentrations at these points and assures that, given time, contaminant breakthrough will occur.

In general, the assumptions outlined above tend to result in conservatively high concentrations of contaminants in the exposure wells. We believe the estimated concentrations could be two or more orders of magnitude greater than actual concentrations.

One additional transport modeling assumption is very important. We have assumed that the earth materials in both our unsaturated and saturated zones are uniform and unfractured, i.e., homogeneous and isotropic (except for dispersivity in the saturated zone). Thus, the analysis does not consider contaminant flow through fractures or discrete zones of high permeability. This type of flow could result in more severe contamination by channeling high concentrations of contaminants rapidly to points of exposure or discharge or conversely, result in very low exposures due to channeling contaminants away from exposure points. We estimate that peak concentrations could easily be an order of magnitude higher than those we have modeled where channeling to exposure points occurs.

2.3 Volatilization - Releases to the Atmosphere

Because many of the constituents of hazardous wastes have appreciable vapor pressures, and can therefore volatilize from disposal facilities, we have recently modified the risk model to include algorithms for volatilization. The resulting emission rates are linked to an atmospheric transport algorithm to allow the model to predict risks through inhalation exposures. We are currently in the process of documenting our approach for estimating volatilization: it is not covered in the appendices.

Volatilization rates are a function of the vapor pressure of the constituent, the concentration gradient between the wastes and the atmosphere, and the resistance to mass transfer that the physical matrix of the waste poses. Several investigators have developed algorithms for estimating volatilization rates under different conditions. We reviewed these algorithms, and selected several for use:

- Thibodeaux and Hwang's model for land farming of petroleum wastes: landfill during operation/no cover;

- Farmer's equation for covered landfills: landfill during operation/daily cover, landfill after closure, surface impoundment after closure; and

- Thibodeaux, Parker, and Heck's steady state model for nonaerated surface impoundments: surface impoundments during operation. We modified this model by using additional equations to predict liquid and gas phase mass transfer coefficients.

For operating landfills, we estimated the proportion of time that disposed wastes are uncovered and are covered by daily fill. The Thibodeaux and Hwang model is used for the uncovered condition, and Farmer's equation is used for the daily fill condition. Farmer's equation is also used (with different inputs for cover thickness, porosity, etc.) to estimate landfill emissions after the cap is installed.

The Thibodeaux, Parker, and Heck model is used to simulate, at closure, emissions during the operating life of surface impoundments. When operations cease, we assume that the wastes are solidified and a cap placed on the impoundment. The model uses Farmer's equation to estimate emissions for the post-closure period.

The model estimates air emissions on a pollutant-by-pollutant basis under each set of relevant conditions. The resulting estimates are scaled to reflect constituent concentration in the waste, the time period for which each set of conditions applies, and facility size (surface area and depth affect emissions rate).

The calculations reflect a number of assumptions and simplifications dealing with factors like mean temperatures, mean wind speeds, topsoil depth and cover moisture content (for landfills and post-closure surface impoundments), and cap material and thickness. Because of the numerous assumptions and simplifications, we believe that these estimates, at best, have order of magnitude accuracy.

2.4 Atmospheric Transport Modeling

The atmosphere can transport volatile waste constituents, leading to inhalation exposures. The model uses a simple algorithm to incorporate atmospheric transport in the risk model. We have not yet completed the documentation of this portion of the model.

The model uses the Atmospheric Transport Model (ATM), a component of the Office of Toxic Substances' Graphic Exposure Modeling System (GEMS), to model annual average ground-level concentrations from hypothetical disposal facilities at ten locations throughout the U.S. Because ATM is primarily a point source model while landfill and surface impoundment emissions are more aptly treated as area sources, we used an adjustment EPA described in an air modeling publication. 1/ In developing the algorithm we analyzed three issues: whether site-specific differences in meteorology result in significant differences in estimated concentration; whether estimated concentration is linearly proportional to emission rate; and whether estimated concentration is sensitive to atmospheric half-life.

The analysis showed that the site-to-site variation in concentrations was relatively small (holding distance from the source constant). Along the wind vector with highest concentrations, the difference between highest and lowest concentration at any of ten of the modeled hypothetical locations was within a factor of four. Differences in concentrations along different wind directions were also relatively small (again, holding distance from the pollution source constant), with the maximum concentration exceeding the minimum concentration by an average factor of about seven. Finally, estimated concentrations were directly proportional to emission rate, and relatively insensitive to atmospheric half-life for all half-lives greater than about ten hours.

Based on these results, and the relative precision of other components of the risk model, we considered it reasonable to estimate ambient concentrations by developing a set of coefficients that linearly relate

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1/ Turner, D. B. 1970. Workbook of atmospheric dispersion estimates. U.S. EPA, Office of Air Programs, RTP, NC. pp. 39-40.

concentration to emission rate. We developed these coefficients for various distances from the source, and use them to estimate annual average ambient air concentrations for populations exposed to emissions of volatile constituents. All of the volatile constituents that we have modeled have atmospheric half-lives longer than ten hours, so we have not developed any chemical-specific coefficients. The coefficients are listed below.

Distance from source (km)	Concentration/emissions coefficient (mg/m^3 per kg/day of emissions)
0.5	1.3 E-05
1.5	3.5 E-06
2.5	1.5 E-06
3.5	8.5 E-07
6.5	3.3 E-07
11.5	1.0 E-07

2.5 Estimating Exposures

Estimating the exposures from ingesting contaminated ground and surface water and from inhaling polluted air, requires basic information on the characteristics of the drinking water sources and the size and distribution of the exposed population. A detailed discussion of the exposure methodology is provided in Appendix C, and is summarized below:

Exposure to a chemical substance is a function of its concentration in an exposure medium, the intake rate of this medium, and the size and demographic characteristics of the exposed population.

For drinking water exposures, the model estimates concentrations in one of two ways. For groundwater, we assume the concentrations output from the subsurface transport submodel (scaled by contaminant loading rate) represents the untreated well water concentrations. For surface water, we assume the contaminated groundwater plume discharges completely into the surface water. Then, the load that the subsurface transport submodel predicts dilutes in the mean annual flow of the surface water; there are no degradation or partitioning processes downstream.

In both cases, we currently assume that there is no subsequent treatment, or that it is ineffective in removing any of the constituent (although current work on modifications incorporate the capability to detect and treat contaminated water). When several sources supply the drinking water we model dilution by the other (presumably uncontaminated) sources by simply multiplying the estimated concentration (from the submodel) by the ratio of water supplied from the contaminated source to the total water supply.

The size of the population exposed to the contaminated drinking water is equal to the number of people served by the water supply. The number of exposed individuals (receptors) is needed to calculate population risks (e.g., number of excess cases expected). Because the risk model covers a time period of 400 years, some estimate of population as a function of time for this period is necessary. We assume people live seventy years, and that the population remains static.

The intake rate of drinking water is assumed to be 2 liters per day, in accord with standard EPA assumptions.

Except for the intake rate, all of the information needed to address the drinking water exposure components is site-specific. This information includes the prevailing groundwater flow direction, locations of wells or surface waters that could be affected by contamination from a disposal facility, mean annual flow of surface waters (if applicable), number of other water sources comprising the water supply, and the number of people served by the water supply over time.

The model has the ability to calculate risks under different assumptions about the size of the exposed population: the site-visit input data set uses the best available information on groundwater use for drinking water; the "augmented" input data set uses the actual population density around disposal facilities and assumes that all people within some distance (usually 5 km) are exposed; and the "hypothetical" input data set assumes that 1000 people are exposed. Population mobility (i.e., changes in location over a diurnal or longer period) is assumed to be negligible.

For inhalation exposures, the relevant concentration of pollutant is determined directly from the atmospheric transport algorithm. We do not account for background concentrations of pollutants, or possible differences in indoor and outdoor concentrations.

The model can also use different assumptions about the size of the population exposed to air emissions, but generally determines the actual population density around disposal facilities and assumes that all people within some distance (usually 5 km) are exposed. As for ingestion exposures, population mobility is assumed to be negligible.

We assume that receptors inhale about 22 cubic meters of air per day. This is in general agreement with inhalation rates used in other EPA studies, which vary from about 10 to 40 cubic meters per day. The chosen value appears to have a minor effect on the results.

Finally, we assume that the average weight of receptors is 70 kg for both air and drinking water exposures. This represents an average weight for an adult. Because most of the receptors' 70-year lifetime is spent as an adult, we used an adult weight rather than a composite weight. The assumptions on receptor weight, however, have little effect on the results compared to other parameters in the risk model.

2.6 Risk and Health Effects Modeling

Estimating human health effects entails predicting quantitatively the consequences of human exposure to disease-causing agents. Traditionally, toxicologists use two different approaches to characterize these health effects. First, they commonly accept a non-threshold assumption for carcinogenic materials. Policymakers have followed their lead, using a linear, non-threshold dose-response model to calculate a unit risk value (risk per unit dose) for a chemical at low doses which, in turn, enables them to calculate explicit risks to individuals and case counts. Second, they assume that there is a threshold for materials that are not carcinogens. For these materials, a linear low dose-response model is not generally used to estimate human health risks. Rather, policymakers usually divide an experimentally determined threshold by a safety factor to estimate an allowable daily intake (ADI) for threshold pollutants, instead of estimating an explicit individual risk.

The hazard submodel diverges from the traditional approach for threshold pollutants because the binary nature of the traditional approach is not very useful for explicit cost/risk analysis. We have developed a more general hazard model which estimates risk from both carcinogens and noncarcinogens as a continuous function of dose, enabling us to treat risks in a fashion more amenable to analysis. In the following sections we describe this hazard submodel and discuss the methodology for developing each health effects parameter. Detailed information concerning the development of the submodel appears in Appendix D, Risk Estimation Methodology.

2.6.1 Model Description

Under this submodel, risk (R) to an individual is a function of dose (D), the shape of the dose response curve (k), the probability of harm or the risk per unit dose (H), and a scaling factor (S) that allows comparisons of graded and dichotomous effects. Specifically:

$$R = (H) \times (S) \times (D)^k$$

Multiplying this risk by the size of an exposed population estimates the risk to that population.

Estimating H, S, and k. We define inherent hazard (H) as the probability of a given response per unit dose of a chemical; this is equivalent to the definition of unit risk EPA uses to describe the potency of carcinogenic materials. Therefore, for carcinogens, the model uses unit risks that EPA has derived. These estimates are based on the assumption that the dose-response relationship is linear, has no threshold, and can be extrapolated from experimental data by using a linearized multistage model. The slope of the resultant dose-response curve is independent of dose throughout the low-dose range and is equal to the inherent hazard of the carcinogen. When EPA values were not available, the model used a simple approximation using the one-hit model to estimate unit risk.

In most cases, the data defining these dose-response relationships are from animal studies. To use these data to estimate human risk we have assumed that the risk to humans is equal to the risk to animals at the same dose per surface area. This assumption parallels traditional EPA policy for carcinogenic risks. It is important to remember in interpreting results that EPA estimates of potency are based on (1) a conservative specification of the dose-response curve and (2) the upper 95 percent confidence limit of the slope of the dose response curve (potency).

There are no comparable EPA unit risk values for noncarcinogens, so we developed estimates for these chemicals. The submodel assumes that response declines as a function of dose[1] to the k^{th} power in the low dose region. Based on this relationship, H can be derived from the experimental minimum effective dose (MED) and the frequency response at that dose (R_0) according to the following equation:

$$H = R_0/(MED)^k$$

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[1] For noncarcinogens, we consider human dose equivalent to animal dose on a dose per body weight basis (rather than dose per surface area).

We can obtain R_0 and the MED from the experimental toxicology literature for each noncarcinogen. Although it is preferable that these values represent chronic effects, frequently it was necessary to adjust the results from subchronic studies using decision rules to derive chronic MEDs (see Appendix D).

The experimental factors R_0 and MED have different meanings depending on the type of effect being studied. Hence, we must approach the calculation of H for dichotomous responses differently than if the type of response were a graded response.

If the response is dichotomous then the derivation of H is fairly straightforward using the method described above. R_0, in these cases, is the frequency of response in the experimental population at the MED, and is usually equal to about 10 percent (the smallest statistically significant detectable response in most experiments of practical size). When both of these values are available we can calculate H simply as $R_0/(MED)^k$. When R_0 is not available we assume that it is equal to 10 percent.

When we derive H from data involving a graded response (such as an enzyme level or body weight change) it is often not possible to determine the incidence of effects. In these cases we conservatively define the MED as the dose at which all the individuals in an experimental population are minimally affected, unless otherwise indicated. This assumption serves to "convert" the graded response data to a dichotomous variable and allows us to calculate H. Since we are assuming that the response (i.e., the fraction of experimental population minimally affected) declines as $(D)^k$ declines, H for graded response effects is equal to $1/(MED)^k$.

To relate risk scores for graded responses to scores for dichotomous responses the model uses a scaling index of 0.1 for relatively nonserious effects, and 0.5 for more serious effects. The scaling factor for birth defects and carcinogens is 1.0. These factors account for conservative assumptions in estimating H values for graded responses.

The shape of the dose-response curve reflects the probability of a toxic response as a function of the dose of the chemical producing this response. In the analysis, we estimate risks at doses well below those administered in experiments. To do this we assume that a linear relationship exists for carcinogens at low doses, resulting in k = 1 for these chemicals.

As previously mentioned, there is no compelling theoretical reason for assuming a linear dose-response for noncarcinogenic materials at low doses. It does appear, however, that in regions below the range of doses for which data are available for noncarcinogens, the frequency (or intensity) of a response decreases relatively quickly as dose decreases.

To determine the k values for noncarcinogens, EPA contracted Science Research Systems to collect and statistically analyze a number of data sets obtained from primary literature. The types of responses analyzed included teratogenic and reproductive toxicity and other noncarcinogenic effects. The k values for the data sets ranged from 1.70 to 2.26, with a median value of 1.74. 1/ With three exceptions, we chose k = 2.00 as a reasonably close approximation of the median value to describe the dose-response curve for all noncarcinogens. To avoid overestimating the risks for lead, copper, and fluoride, we chose k = 4 because available human data indicate that these risks decline very rapidly at doses below the MED.

2.6.2 Results

Values for inherent hazard, severity scaling, and the dose-response curve shape for the 96 chemicals in the liner/location data base are listed in Figure 2-7.

It is important to note that the risk equation we developed predicts no biological threshold. Based on the theoretical assumption that exposure to

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1/ RCRA Risk-Cost Analysis Model, Phase III Report. Appendix F, Dose-Response Curves for Non-Cancer Effects. March, 1984. ICF Incorporated. A full text of this study is given in Attachment 1 of Appendix D of this document.

Figure 2-7

TOXICITY PARAMETERS

Constituent	Unit Risk (H)	Scaling Factor (S)	Dose Response Parameter (k)
Acenaphthene	0.1150E+02	1.0	1
Acetaldehyde	0.1590E-03	0.5	2
Acetone	0.8000E-05	0.5	2
Acetonitrile	0.3600E-01	0.5	2
Acrolein	0.6600E+00	0.5	2
Acrylonitrile	0.2400E+00	1.0	1
Aldicarb	0.1000E+02	1.0	2
Allyl alcohol	0.6270E+01	0.5	2
Aniline	0.1900E-01	1.0	1
Antimony	0.1800E-03	0.5	2
Arsenic	0.1500E+02	1.0	1
Barium	0.3000E+00	0.5	2
Bis(chloromethyl)ether	0.9300E+04	1.0	1
Benz(a)anthracene	0.1150E+02	1.0	1
Benz(a)pyrene	0.1150E+02	1.0	1
Benzene	0.5200E-01	1.0	1
Benzotrichloride	0.1210E+02	0.5	2
Benzyl chloride	0.1800E-01	0.5	2
Cadmium	0.7800E+01	1.0	1
Carbon disulfide	0.5300E-01	1.0	2
Carbon tetrachloride	0.1300E+00	1.0	1
Chlordane	0.1610E+01	1.0	1
Chloroacetaldehyde	0.3760E+02	0.5	2
Chlorobenzene	0.2360E-02	0.5	2
Chloroform	0.7000E-01	1.0	1
2-Chlorophenol	0.1000E-01	0.5	2
Chromium VI	0.2000E-01	0.5	2
Chrysene	0.1150E+02	1.0	1
Copper	0.2560E+02	0.5	4
Cyanides	0.1000E+00	0.5	2
Cyclohexane	0.5900E-04	0.5	2
1,2-Dichlorobenzene	0.3400E-03	0.5	2
1,4-Dichlorobenzene	0.4300E-03	0.5	2
1,2-Dichloroethane	0.6900E-01	1.0	1
1,1-Dichloroethene	0.1470E+00	1.0	1
Dichloromethane	0.4500E-02	1.0	1
12-Dichloropropane	0.3900E-01	1.0	1
Dichloropropanols	0.3900E-01	1.0	1
13-Dichloropropene	0.9400E-01	0.5	2
Dimethoate	0.5310E+01	0.1	2
Dimethyl alkylamines	0.4050E-02	0.5	2
1,3-Dinitrobenzene	0.8860E-01	0.1	2
2,4-Dinitrotoluene	0.3100E+00	1.0	1
Dinoseb	0.3200E-03	1.0	2
Endosulfan	0.3000E-02	0.5	2
Epichlorohydrin	0.9900E-02	1.0	1
Ethylbenzene	0.5760E-03	0.5	2
Ethylene oxide	0.6300E+00	1.0	1

Figure 2-7 (Continued)

TOXICITY PARAMETERS

Constituent	Unit Risk (H)	Scaling Factor (S)	Dose Response Parameter (k)
Fluoride	0.1500E+08	0.5	4
Formaldehyde	0.7700E+00	0.5	2
Hexachlorobenzene	0.1670E+01	1.0	1
Hexachlorobutadiene	0.7750E-01	1.0	1
Hexachlorocyclopentadiene	0.1000E+07	0.1	2
Hexachloroethane	0.1420E-01	1.0	1
Hydroquinone	0.2500E+00	0.5	2
Lead	0.1250E+10	0.1	4
Lindane	0.1330E+01	1.0	1
Methyl,ethyl Ketone	0.1100E-06	1.0	2
Methyl,isopropyl Ketone	0.4480E-03	0.5	2
Maleic Anhydride	0.2780E-05	0.5	2
Mercury	0.6900E+00	0.5	2
Methomyl	0.6940E-02	0.5	2
Methyl chloride	0.9850E-03	0.5	2
Methyl isocyanate	0.7900E+01	0.5	2
Methyl methacrylate	0.2500E-04	0.1	2
Molybdenum	0.1480E+03	0.5	2
Naphthalene	0.1000E-03	0.5	2
Naphthoquinone	0.2500E+00	0.5	2
Nickel	0.9000E+01	1.0	2
Nitrobenzene	0.1800E-01	0.5	2
4-Nitrophenol	0.1800E-01	0.5	2
Paraldehyde	0.1590E-03	0.5	2
Parathion	0.6940E+06	0.1	2
Polychlorinated biphenyls (as Ardor 1254)	0.4340E-01	1.0	1
Pentachloronitrobenzene	0.7000E+00	1.0	1
Penthachlorophenol	0.1600E-01	1.0	2
Phenol	0.1000E-01	0.5	2
Phthalic Anhydride	0.4000E-06	0.5	2
Pyridine	0.6600E-01	0.5	2
2,3,7,8-Tetrachlorodibenzo-diozin	0.1600E+06	1.0	1
Toluene diisocyanate	0.4300E-02	1.0	1
1,1,1,2-Tetrachloroethane	0.8100E-02	0.5	2
1,1,2,2-Tetrachloroethane	0.2000E+00	1.0	1
Tetrachloroethylene	0.3500E-01	1.0	1
Thallium	0.3090E+02	0.5	2
Toluene	0.9400E-03	0.1	2
Toluene-2,4-Diamine	0.2900E+00	1.0	1

Figure 2-7 (Continued)

TOXICITY PARAMETERS

Constituent	Unit Risk (H)	Scaling Factor (S)	Dose Response Parameter (k)
Toxaphene	0.1130E+01	1.0	1
1,2,4-Trichlorobenzene	0.2000E-01	0.5	2
1,1,1-Trichloroethane	0.1600E-02	1.0	1
1,1,2-Trichloroethane	0.6800E+00	1.0	1
Trichloroethylene	0.1900E-01	1.0	1
Vanadium	0.3000E-01	0.5	2
Vinyl Chloride	0.1750E-01	1.0	1
Xylene	0.2200E-02	0.5	2
Zinc	0.1000E-03	0.5	2

carcinogenic substances results in a finite response at all finite doses, this equation describes the risk from carcinogens. However, the no-threshold prediction is not consistent with traditional toxicological theory.

2.6.3 Limitations and Assumptions

The primary limitations and assumptions in the hazard submodel are those traditionally found in determining inherent hazards and characterizing dose-response in the low dose regions. These specific and traditional sources of uncertainty are:

- Considering human and animal toxic responses to be similar at similar doses. Differences in size, lifespan, metabolism, and physiology between humans and test animals complicate the applicability of animal data to estimates of human risk. However, the scientific community generally accepts animal responses as a valid indicator of human responses.

- Assuming that the population at risk is a uniform population of healthy adults with a life span of 70 years. This prevents us from accounting for intraspecies differences in response to hazardous exposures.

- Assuming that experimental data from one route of exposure can be used to assess toxic effects via another route when the effect is distant from the point of entry into the body. (For example, we consider the liver toxicity of a compound to be equivalent via inhalation exposure or via ingestion.) However, we do not use this assumption when the effect is at the point of entry, i.e., we do not use ingestion toxicity data to assess the respiratory toxicity of a compound, since this is likely to be directly related to an inhalation route of exposure.

- Using data from subchronic studies to infer chronic toxic effects. Because the hazard submodel assesses human health risks due to chronic lifetime exposures, data from long term studies are preferable. Often these data have not been available so we have extrapolated from subchronic studies which used relatively high experimental doses. This increases the uncertainty in our assessment of chronic potencies and creates the possibility of either ignoring an effect that occurs only after chronic exposures or considering an effect that is the result of doses considerably higher than those encountered under environmental conditions.

- Assuming that the dose-response for carcinogens is linear at low doses and that there is no threshold dose. This theoretical approach is widely accepted and often used by policymakers to provide a conservative estimate of carcinogenic risk.

- Assuming that response from exposure to noncarcinogens at low doses declines more rapidly with decreasing dose than a linear function would predict. The use of relatively small safety factors to regulate exposures to noncarcinogens by several regulatory agencies, which implies that the dose-response is a power function rather than linear, is consistent with this assumption. Results of our examination of dose-response data for a wide range of noncarcinogens concur.

- Assuming an arbitrary risk threshold for noncarcinogens. A large amount of experimental evidence indicates that noncarcinogens exhibit threshold effects. The risk algorithm assumes nonthreshold effects so the model incorporates the assumption of noncarcinogen thresholds by considering risks below 10^{-5} (10^{-9} for lead) for noncarcinogens as negligible.

- Comparing risks from chemicals which elicit fundamentally different adverse responses. The complexity in ranking different toxic effects according to the severity of the response has created difficulties in developing a universally accepted severity index. We have developed only a very limited scaling system to allow a rough comparison of the magnitude of risks for different effects when using the model.

2.7 Computer Structure of Risk Model

A single computer framework integrates the model components covering liner failure, subsurface transport, volatilization, atmospheric transport, exposures, and dose-response relationships. The most complicated submodels (failure analysis, saturated zone transport, volatilization, and atmospheric transport) are incorporated as summarized output tables rather than as computational models. The user specifies input variables, and the model automatically retrieves the appropriate values from the summary tables and scales them depending on the size of the facility and other variables. The other algorithms are programmed into the model as continuous functions. More complete descriptions of the computer structure of the risk model can be found in Appendices C and G, along with the major simplifications and assumptions. The computer structure is summarized below.

The releases from the facility are routed to the appropriate components: volatile emissions are input to the atmospheric transport submodel; and leachate mass loadings are input to the unsaturated zone submodel, which subsequently links to the saturated zone submodel. A mass balance routine allocates waste mass to the two media until the mass is depleted. The model calculates up to 400 years of exposure for releases to air and up to 400 years of exposure for releases to groundwater (up to 200 years of release plus up to 200 years for the released pollutant to reach a well). In many cases modeled so far, all of the waste mass is depleted by simulation year 200.

The model then calculates the 70-year running average exposures for each medium. These running average exposures are combined with the chemical-specific toxicity factors (S, H, k) to calculate individual and population risks on a yearly time-step.

When summarizing the risks for facilities, the model can account for the fact that different pollutants pose different types of health risks in four ways:

(1) List the risk for each effect at a facility separately.

(2) List cancer risks separately but group all other effects.

(3) Group all effects.

(4) Develop and use a weighting scheme to assign different severity values to different effects, based on judgments of relative quality of life lost.

Most of the applications to date have employed the second and third options.

The risk model output is stored in a SAS file for statistical analysis. We have developed a number of tabular and graphical outputs to display summary data on risk results and intermediate results (e.g., year of first release, cumulative release rates, years of exposure, maximum exposure, and time delay in the unsaturated zone). Chapter 4, Applications of the Model, provides some examples.

3. THE COST MODEL AND COST/RISK ANALYSIS

This section of the report summarizes how the model calculates costs for landfills and surface impoundments of differing sizes and designs, and the results of these calculations; Appendix E contains a more complete description. To calculate costs, the project team specified designs and sizes, estimated the engineering costs for these designs and sizes, and converted the engineering costs to economic measures by discounting. The results show that there are substantial returns to scale in land disposal--the unit disposal cost decreases as the facility size increases--and that the cost of land disposal increases as the design of the facility becomes more stringent.

3.1 Engineering Costs

OSW's Facilities Design Computer Cost Model, which estimates construction and closure costs for landfills and surface impoundments of different liner configurations and varying sizes, generated the costs for all the designs specified in the failure model. For applications, we used the designs shown in Figure 3-1. For landfills and surface impoundments, both the unlined and clay-lined designs comply only with Part 265 RCRA design requirements. The single synthetic design complies with Part 264, while the double synthetic, synthetic/clay, and double synthetic/clay designs not only comply with Part 264, but also include features that RCRA does not currently require.

The cost model simulates the significant design and engineering components of facility construction and estimates their cost, often called "engineering cost," using data from standard building, construction, and site work cost manuals. The model estimates initial capital costs (land, constructing the first year's cell for landfills or excavating and installing liners for surface impoundments, building the infrastructure, etc.), annual capital costs (building the next year's cell and closing the

Figure 3-1

LANDFILL AND SURFACE IMPOUNDMENT DESIGNS
A. Landfill Designs

1 Single Synthetic	2 Double Synthetic	3 Single Clay	4 Synthetic/Clay	5 Double Syn/Clay	6 Baseline-Unlined
			FINAL COVER		
Revegetation	Same as Design 1	Same as Design 1	Same as Design 1	Same as Design 1	Revegetation
0.6 m select fill layer					0.15 m select fill layer
Geotextile filter layer					
0.3 m sand layer					
30 mil syn					
Geotextile support layer					
0.6 m clay layer					
			CONTAINMENT SYSTEM		
0.15 m earth fill layer	0.15 m earth fill layer	0.15 m earth fill layer	0.15 m earth fill layer	0.15 m earth fill layer	None
Geotextile filter layer	Geotextile filter layer	Geotextile filter layer	Geotextile filter layer	Geotextile filter layer	
0.3 m sand layer	0.3 m sand layer	0.3 m sand layer	0.3 m sand layer	0.3 m sand layer	
36 mil syn	36 mil syn	0.6 m clay layer	36 mil syn	36 mil syn	
Geotextile support layer	0.3 m sand layer		Geotextile support layer	0.3 m sand layer	
	36 mil syn		0.6 m clay layer	36 mil syn	
	Geotextile support layer			Geotextile support layer	
				0.6 m clay layer	

Figure 3-1

LANDFILL AND SURFACE IMPOUNDMENT DESIGNS

B. Surface Impoundment Designs

1 Single Synthetic	2 Double Synthetic	3 Single Clay	4 Synthetic/Clay	5 Double Syn/Clay	6 Baseline-Unlined
			FINAL COVER		
Revegetation 0.6 m select fill layer Geotextile filter layer 0.3 m sand layer 30 mil syn Geotextile support layer 0.6 m clay layer	Same as Design 1	Same as Design 1	Same as Design 1	Same as Design 1	Revegetation 0.15 m select fill layer
			CONTAINMENT SYSTEM		
36 mil syn	36 mil syn 0.3 m sand layer 36 mil syn Geotextile support layer	0.6 m clay layer	36 mil syn 0.3 m sand layer 0.6 m clay layer	36 mil syn 0.3 m sand layer 36 mil syn Geotextile support layer 0.6 m clay layer	None

last year's cell for landfills), closure costs (covering the final cell for landfills or covering the surface impoundment, decontaminating equipment, monitoring, and certifying closure), and operating and maintenance (O&M) costs associated with the technical aspects of operating the facility. For surface impoundments, it also estimates the frequency and cost of dredging. The model does not include O&M costs for recordkeeping, reporting, and financial responsibility because these Part 264 requirements do not change as facility designs change. Thus, the costs underestimate the total actual costs of disposing of hazardous waste, but are valid for performing incremental cost/risk analyses.

There are several important assumptions underlying the engineering cost estimates. First, we assumed that facilities are able to use on-site clay in all designs requiring clay. If they are not able to do so, engineering costs would be larger, ranging from 12 to 53 percent more than those calculated, depending on facility design and size. Second, we assume that all surface impoundments are on-site, and that smaller landfills are also on-site and do not require the fences, gates, office buildings, scales, and equipment the larger off-site facilities require.

Although the current applications analyze costs for eight sizes of landfills and six sizes of surface impoundments with six liner designs for each size, the cost model is flexible with respect to designs and sizes. Thus, it is easy to add other liner configurations, modify those currently in the model, or explore the costs of other sizes to evaluate the costs of a wider range of regulatory options.

3.2 Economic Measures

The model transformed the engineering costs into measures which facilitate economic analysis. First, it calculates total present values for each facility design and size by discounting the engineering costs in each year of the facility's life and summing the discounted costs over all years. In calculating these present values, we used a three percent real discount rate, which is near the midpoint of the range of rates indicated by both the consumption rate of interest and the social rate of time preference, as indicated in EPA's Guidelines for Preparing Regulatory Impact Analysis. We also used EAB's standard assumptions concerning facility lives and operating procedures: surface impoundments and landfills both operate for 20 years; operators open the first cell in a landfill at the beginning of year one and cover it at the beginning of year two, and cover the last cell at the beginning of year 21; operators line a surface impoundment at the beginning of year one and cover it at the beginning of year 21.

Costs are often unevenly distributed through time, preventing direct comparisons, so after calculating the present values, the model calculates an equivalent annual cost spread evenly over the life of the facility. The model annualizes by multiplying the present value of the costs of each facility size and design by a capital recovery factor. With a three percent real rate of return and a 20 year facility life, the capital recovery factor is 0.0672.

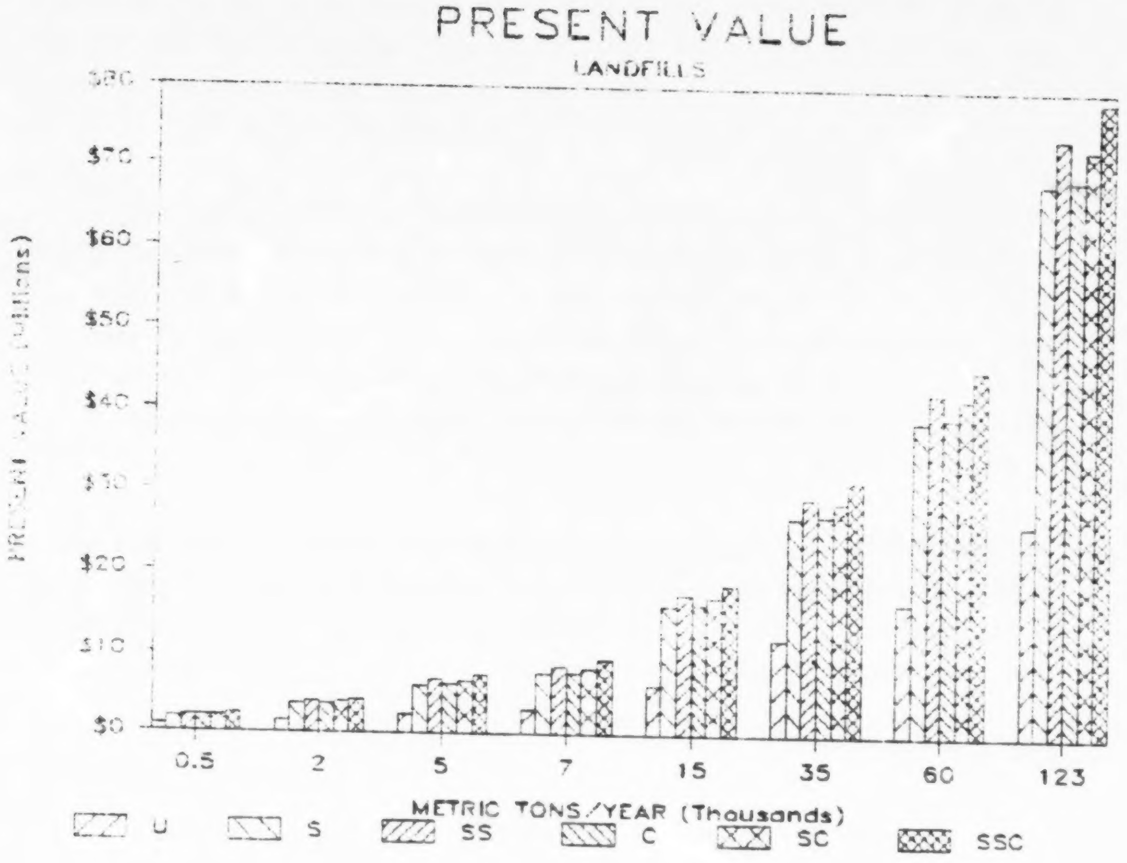
Then the model calculates the annualized present value on a per unit basis. To do so, it divides the annualized cost for each facility size and design by its capacity. This results in annualized costs per ton for landfills, and annualized costs per acre and per gallon for surface impoundments.

3.3 Results

Figure 3-2 shows present values of landfill costs for each of the technology/size combinations. The figure, as expected, shows that the unlined design is less expensive than the lined designs, and double-lined facilities are slightly more expensive than single-lined facilities. It also shows that total costs increase as size increases but at a slightly lower rate. Figure 3-3 provides the same information for surface impoundments and leads to the same conclusions.

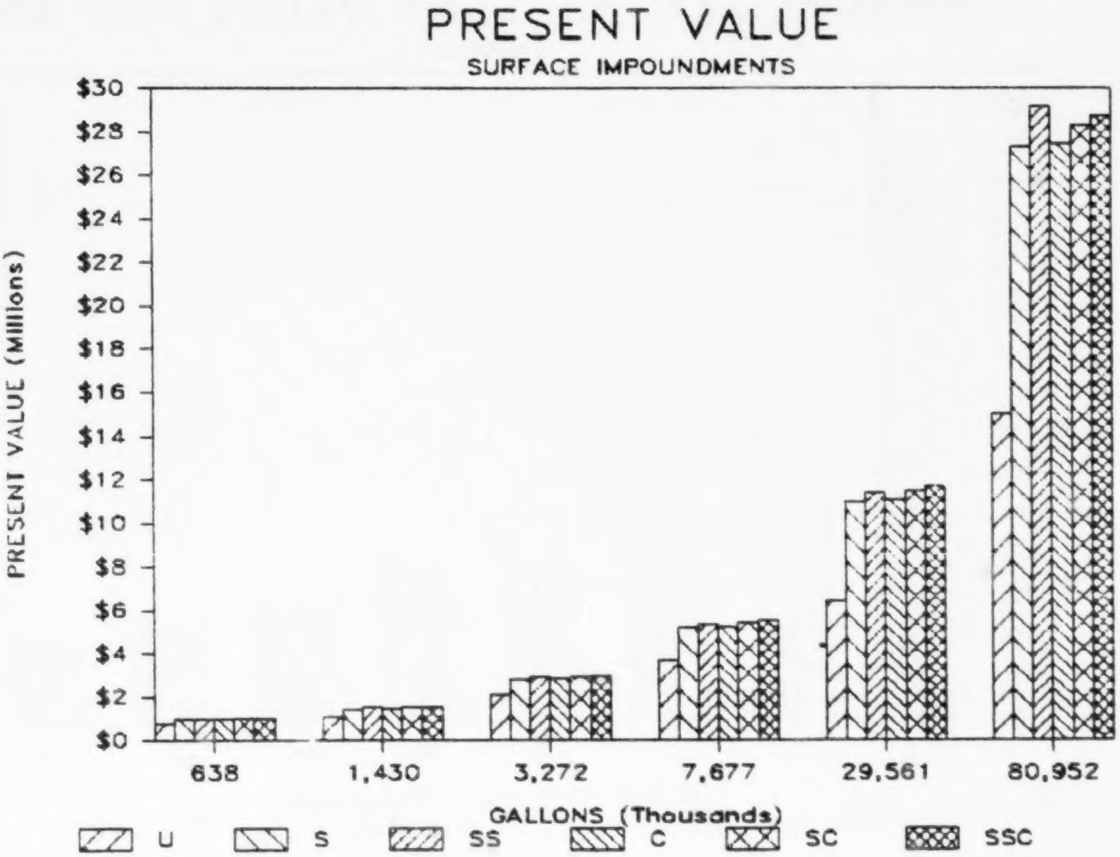
Calculating cost per ton for landfills and cost per gallon for surface impoundments depicts more precisely how size affects unit costs. Figure 3-4 shows the annualized cost per ton for landfills: as the landfill sizes

Figure 3-2



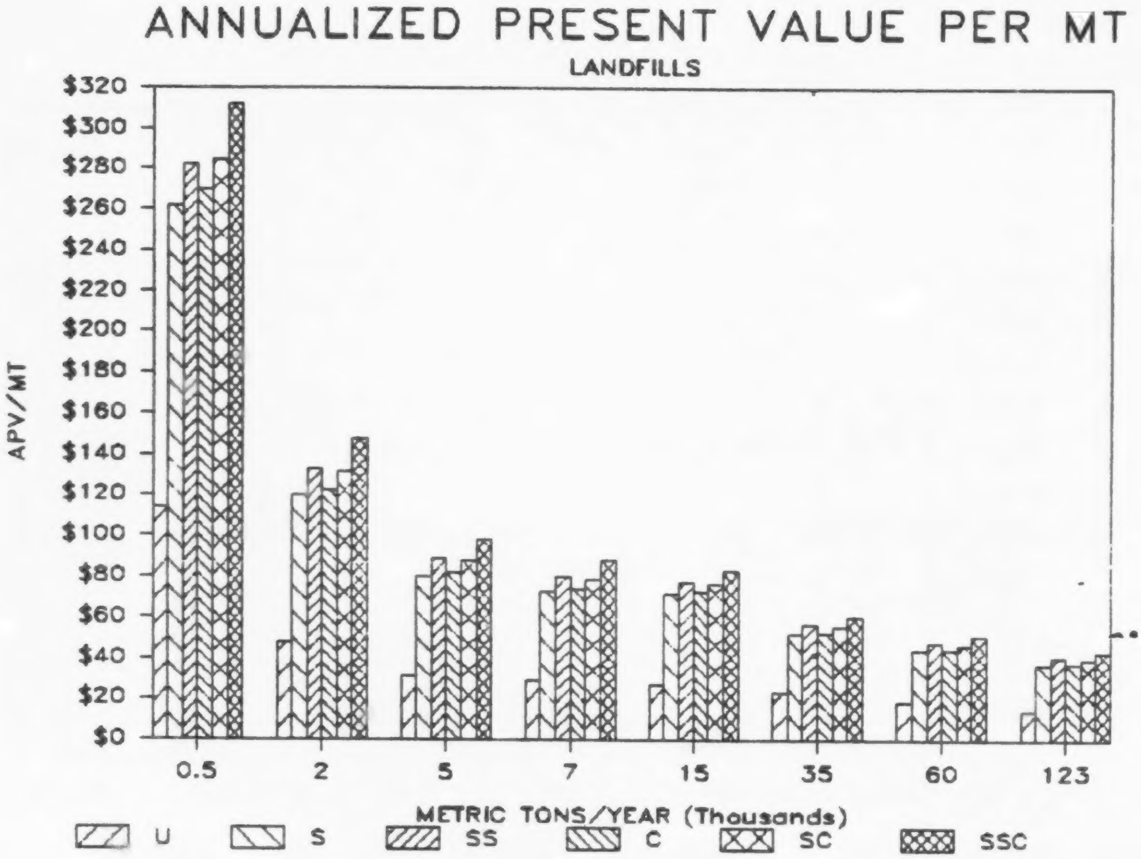
U = Unlined facility
S = Single synthetic liner
SS = Double synthetic liner
C = Clay liner
SC = Synthetic/clay liner
SSC = Double synthetic/clay liner

Figure 3-3



U = Unlined facility
S = Single synthetic liner
SS = Double synthetic liner
C = Clay liner
S/C = Synthetic/clay liner
SS/C = Double synthetic/clay liner

Figure 3-4



U = Unlined facility
S = Single synthetic liner
SS = Double synthetic liner
C = Clay liner
SC = Synthetic/clay liner
SSC = Double synthetic/clay liner

increase, the per unit costs drop dramatically. For example, while the annualized cost for a 2,000 MT per year single synthetic landfill is $120 per ton, the cost for a 123,000 MT per year landfill of the same design is only about $40 per ton. The figure also shows that except for the difference between unlined and lined facilities the differences in the costs resulting from design changes are small when compared to the differences in costs resulting from size changes.

Figure 3-5 shows annualized present values on a per gallon basis for surface impoundments. This figure also shows that returns to scale for surface impoundments are substantial, while cost differences among lined designs are smaller.

The model stores the present values, annual present values, and annual present value on a per unit basis for use in the cost/risk and cost-effectiveness analyses. Finally, for each technology, the model calculates equations for cost as a function of size by fitting regressions to the cost and size data. It stores these cost equations, allowing the user to estimate costs for disposal facilities of additional sizes.

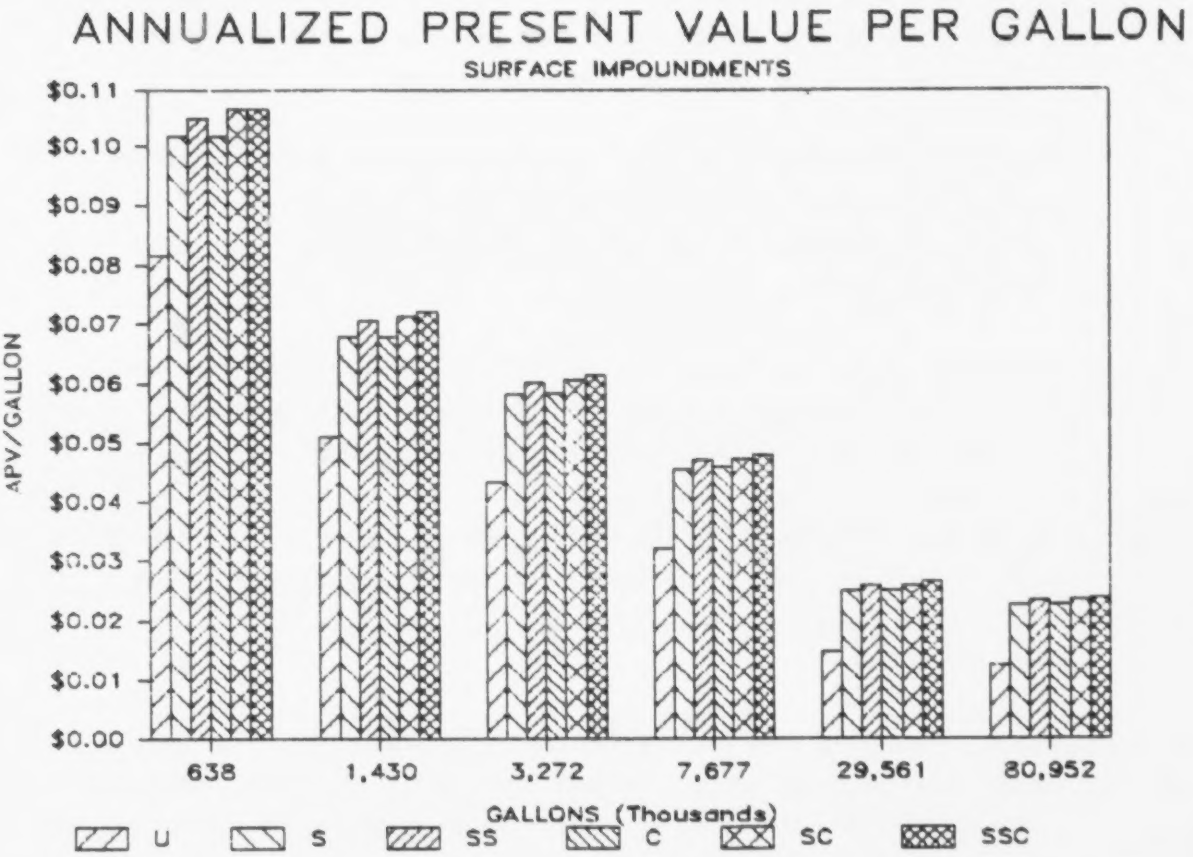
3.4 Limitations

The cost analysis uses engineering cost estimates of facility designs, rather than observed costs at actual facilities. We expect that costs incurred at real facilities will be close to these engineering estimates, but site-specific factors could cause discrepancies between these estimates and actual costs. Further, in annualizing costs, we have assumed that facilities operate for 20 years; we do not have an estimate of average operating life for actual facilities. In spite of the foregoing, these costs are precise enough to provide meaningful cost estimates for cost/risk and cost-effectiveness analyses.

3.5 Cost/Risk Analysis

This section of the report discusses the cost/risk modeling in the Liner Location Risk and Cost Analysis Model.

Figure 3-5



U = Unlined facility
S = Single synthetic liner
SS = Double synthetic liner
C = Clay liner
SC = Synthetic/clay liner
SSC = Double synthetic/clay liner

3.5.1 Methodology

Cost/risk ratios use incremental costs and incremental risks, with present value as the measure of costs and total cases (cancer and noncancer) over a 400 year period as the measure of risk. For example, in calculating the cost/risk ratio associated with requiring a double synthetic liner at a facility instead of a single synthetic liner, we would perform a three-step process. First, we would find the incremental cost by subtracting the present value of the single liner from the present value of the double liner. Next, we would find the incremental risk abatement by subtracting the cases over 400 years with the double liner from the cases with the single liner. Finally, we would divide incremental cost by incremental risk to produce the cost/risk ratio.

The model currently does not discount risk in calculating cost/risk or cost-effectiveness ratios, which implies that EPA is indifferent between averting a case today and averting a case at a distant time. There are, however, two closely related reasons for weighing adverse effects that occur relatively later less heavily than effects of the same magnitude that occur sooner, just as costs that occur later are discounted more heavily:

(1) Individuals and society generally have a positive rate of time preference: they usually prefer a dollar now over a dollar (net of inflation) in the future; they usually prefer to avert a cancer or a death now over doing so in the future.

(2) Adverse health effects such as cancer represent a loss to society that can be expressed in economic terms: loss of income, costs of medical care, etc. When these occur in the future, they should be discounted just like the costs of preventative technology.

Even when the discounting procedure has made costs comparable among scenarios regardless of whether they occur in the near or distant future, we are still more confident about the model's near-term predictions than its long-term predictions. This is because the factors that determine health effects are much less certain over the long run: local populations may grow or decrease; aquifers may become polluted from other causes; and medical advances may provide cures or preventative measures.

3.5.2 Computational Methods

We perform the cost/risk and cost-effectiveness modeling using a simple computer spreadsheet program. Input variables are (1) present values of design costs for the facility configurations and sizes (from the cost model), and (2) risk expressed in terms of cases for either real or hypothetical facilities (from the risk model). The computer program ranks the options in order of increasing protection, and then finds for this ranking the associated incremental costs and risks.

4. APPLICATIONS OF THE MODEL

This chapter describes three applications of the Liner Location Risk and Cost Analysis Model. The purpose of these applications is to demonstrate how the model has been used and to illustrate how it could be used in the future for policy analysis. In the first application, we evaluated national technology requirements from a cost/risk perspective. In the second, we compared the risks of land disposal in different environmental settings, holding waste constituents, quantities, and technology constant. In the third application, we compared the risks of different wastes, holding technology and environmental setting constant. In each application, we used different data sets as input to the components of the risk model; the following describes the three applications and their results.

4.1 Evaluating National Technology Standards

In this application of the Model, we evaluated the costs and risks of requiring national liner standards, using two different sets of input data. The first set represents a cross-section of actual land disposal facilities. The resulting 400 year average risks were very small, and largely driven by our data on receptors. Liners were not very effective in reducing risks and cost/risk ratios for imposing more stringent designs were very high. We therefore performed the risk simulation again using additional landfill sites and assuming that receptors exist at all sites; under this assumption, risks were far larger, liners were somewhat more effective in reducing risks, and cost/risk ratios were very low.

4.1.1 Actual Facilities

We ran simulations for 17 landfills and 31 surface impoundments. Modelled designs of these facilities were taken from the OSW site visit survey data base. This survey provides relatively complete site data and we believe that they comprise a reasonably representative sample of landfills and surface impoundments; it is not, however, a statistically valid random sample.

We attempted to use as much actual data as possible for waste streams, capacities, environmental conditions, distances to wells, and exposed population. We simulated the costs and risks at each facility using hypothetical liner systems.

The site-visit survey obtained data on the ten wastes disposed of in largest volumes at each site. In some cases, respondents identified specific waste constituents as well as their concentrations in the wastes. In these cases, we used the site-specific data to characterize wastes present. Otherwise, we used generic constituents of concern and concentrations from the OSW W-E-T Model for D-, F-, and K-wastes; and used the listed constituent and engineering judgment to estimate concentrations for U- and P-wastes.*

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*Part 261 of RCRA identifies and lists hazardous wastes. Under this Part, wastes may be classified as hazardous based on several criteria:

- "D" wastes are ignitable, corrosive, reactive, or contain specific constituents for which the Agency has issued drinking water standards and which leach from the wastes under specified conditions.
- "F" wastes are listed from nonspecific sources (F007 is spent plating bath solutions from electroplating operations).
- "K" wastes are listed from specific sources (K048 is DAF float from the petroleum refining industry).
- "U" wastes are toxic discarded commercial chemical products, off-specification products, containers, and spill residues.
- "P" wastes are acutely toxic discarded commercial chemical products, off-specification products, containers, and spill residues.

We excluded some reported wastes from our analysis. Although D001, D002, and D003 (ignitable, corrosive, and reactive) wastes comprise a large portion of the wastes disposed of at many of the sites, we excluded them because they have no generally identifiable chronically toxic constituents. We also excluded wastes for which no dose-response functions were available. Finally, we excluded liquid wastes reported in landfills; the surveys are based on 1981 conditions, before EPA placed restrictions on disposing of these wastes in landfills. At sites where we excluded wastes, we still used the reported capacity for costing purposes, which could distort the cost/risk relationships.

We assumed that all of the facilities start operation in simulation year zero, and for the next twenty years dispose of the wastes they reported at the rate they reported. Using these assumptions, our simulations covered disposal of about 800,000 MT/year in landfills and 17,000,000 MT/year in surface impoundments. This accounts for about 26 percent of the total amount of wastes landfilled and about 13 percent of the total amount of wastes managed in surface impoundments.

We ran the failure submodel for each of six facility designs and five climate types. We identified the appropriate climate type for each of the land disposal sites from the site visit survey data base. Failure probability over time is essentially constant for different facility capacities. The failure analysis indicated that triple liners have failure and release rates that are virtually identical to composite (synthetic/clay) double liners; therefore, we did not run separate risk or cost/risk results for the triple liners. We ran the exposure/risk and cost models for each design for each of the sites, so the actual design currently in place at the sites has no bearing on the analysis.

For some sites, we characterized unsaturated zone thicknesses using values from a report by GCA Corporation, Selecting a Representative Sample of Existing Hazardous Waste Land Disposal Facilities Based on Geography, Geology, and Hydrology. (GCA Co., OSW (USEPA 1982)). For unsaturated zone thickness at the remaining sites and for hydrogeologic flow fields we selected values based on regional geographic conditions. Other hydrogeologic parameters (e.g., fraction of organic carbon and porosity) were held constant across all sites. At most sites, we identified several possible flow fields and simulated risks under all of them. Then, for the cost/risk analysis, we used the median risk results over the set of flow fields.

For the air modeling, we estimated volatilization rates for each of the waste streams, in each of the climates, for both landfills and surface impoundments. Then we used a linear scaling factor to translate emission rates to air concentrations at two distances from the sites. For risk to the most exposed individual, we used concentrations at 0.5 km, the closest distance we could model using ATM. For the average exposure to the entire population exposed to air emissions, we judgmentally selected a distance of 3.5 km. We obtained data on the size of the exposed population by reviewing 1980 census data in the OTS GEMS population file for each of the sites.

We collected data on the locations of wells near the facilities, obtaining most of the data on drinking water sources by contacting U.S. Geological Survey offices and state offices; additional sources include computer data bases, public records, publications, and contact with local sources. We also determined the likely direction of groundwater flow. We assumed that wells were potentially at risk if (1) they are located downgradient of the source within an arc of 45° on either side of the likely flow direction, and (2) they are within two miles of the waste facility. We determined the size of the potentially exposed populations from the same data sources as for well locations.

The toxicological factors used to estimate risks for the Liner Location Risk and Cost Model are identical to those in the W-E-T Model.

We calculated the costs of the different liner designs at the sites by mapping the amount of waste in the landfill or surface impoundment into the closest model size we costed. For this analysis, we assumed that liquids were placed into surface impoundments, so that all closure costs include solidifying the waste. We included costs for installing groundwater wells except for designs with double synthetic liners since they are currently exempted from groundwater monitoring requirements. We did not include any administrative costs required under RCRA except for waste testing analyses. For surface impoundments, we assumed that facilities would comply with requirements for additional liners by closing their old surface impoundments and building new ones with the requisite design, rather than retrofitting existing impoundments since this close-and-construct approach is less expensive than retrofitting. We did not include the costs of post-closure care for old impoundments which would occur during the facility lifetime; these costs are relatively minor compared to the original construction costs.

Our simulation using these input data shows the aggregate number of carcinogenic and non-carcinogenic cases is very small regardless of liner requirements, as Figure 4-1 shows. The primary reason for this is that only two of the landfills and five of the surface impoundments have nearby populations using ground water as a source of drinking water. Of these, only one landfill and three impoundments showed any risk, even for the unlined design.

Not only were aggregate cases low, but they were also relatively insensitive to the facility design, as Figure 4-1·shows. For the landfill and for two of the three impoundments, aggregate cases were identical regardless of design, and were less than one case for 400 years. For the third surface impoundment, more stringent liner requirements reduced risk, yet the unlined design resulted in less than one case every 400 years.

Because risks are so low, cost/risk ratios are extremely high. For landfills, additional liner requirements provide no incremental risk abatement, and the cost/risk ratio is undefined. Liners reduce aggregate risk at only one of the 31 surface impoundments. Requiring single synthetic liners as opposed to requiring no liner yields a cost/risk ratio of about $150 million per case avoided. Moving from a single synthetic

Figure 4-1

AGGREGATE RISK AT LAND DISPOSAL FACILITIES BY TECHNOLOGY

(Cases over 400 years)

LINER TECHNOLOGY	FACILITY DESIGN			
	SI1	SI2	SI3	LF1
Unlined	6E-1*	2E-6	2E-1	7E-3
Single Synthetic	2E-2	2E-6	2E-1	7E-3
Single Clay	2E-1	2E-6	2E-1	7E-3
Double Synthetic	1E-2	2E-6	2E-1	7E-3
Synthetic/Clay	1E-2	2E-6	2E-1	7E-3

* Includes 9E-3 cancer risk. The remaining risk in this case, and the total risk in all other cases, consists of noncancer risks.

requirement to a double synthetic requirement costs about $1 billion per case avoided.

4.1.2 Actual Land Disposal Facilities with Hypothetical Populations

Because our results at these facilities were driven so strongly by our data on receptors, we modified the analysis to create scenarios where there was a higher potential for drinking water exposure. First, in this analysis we increased our sample of landfills by using data from OSW's National Survey of Hazardous Waste Generators and Treatment, Storage and Disposal Facilities Regulated under RCRA in 1981 (RIA mail survey). The landfill component of this survey contains detailed information on 79 landfills. After excluding landfills if the responses were confidential, or if we were unable to perform risk modeling because we could not characterize the waste or determine climate or groundwater flows, we were able to simulate risks for an additional 44 facilities. Thus, we modeled 61 landfills and 31 surface impoundments. Second, and crucial to our risk results, we assumed that people living near all of the sites used ground water as their only drinking water source.

We assumed that there is a municipal well 600 meters directly down gradient of each facility, and that the well serves 25 percent of the people within 20 km of the facility. We also assumed all people living within 5 km of the site are exposed to air emissions. We obtained population density data from the ATM-SECPOP Model, and calculated the number of people potentially exposed at each site through the air and groundwater exposure pathways.

For the additional 44 landfills not covered in the first application, we collected data on waste volume and type from the RIA mail survey data base. We characterized wastes and constituents using the same methods as before.

We estimated unsaturated zone thicknesses and groundwater flow fields for each of the additional sites. As in the first application, where there were several potential flow fields for a facility, we modeled risks for all of them and selected the median risk for further cost/risk analysis. All other inputs and assumptions were identical to those we used to model actual facilities with actual populations.

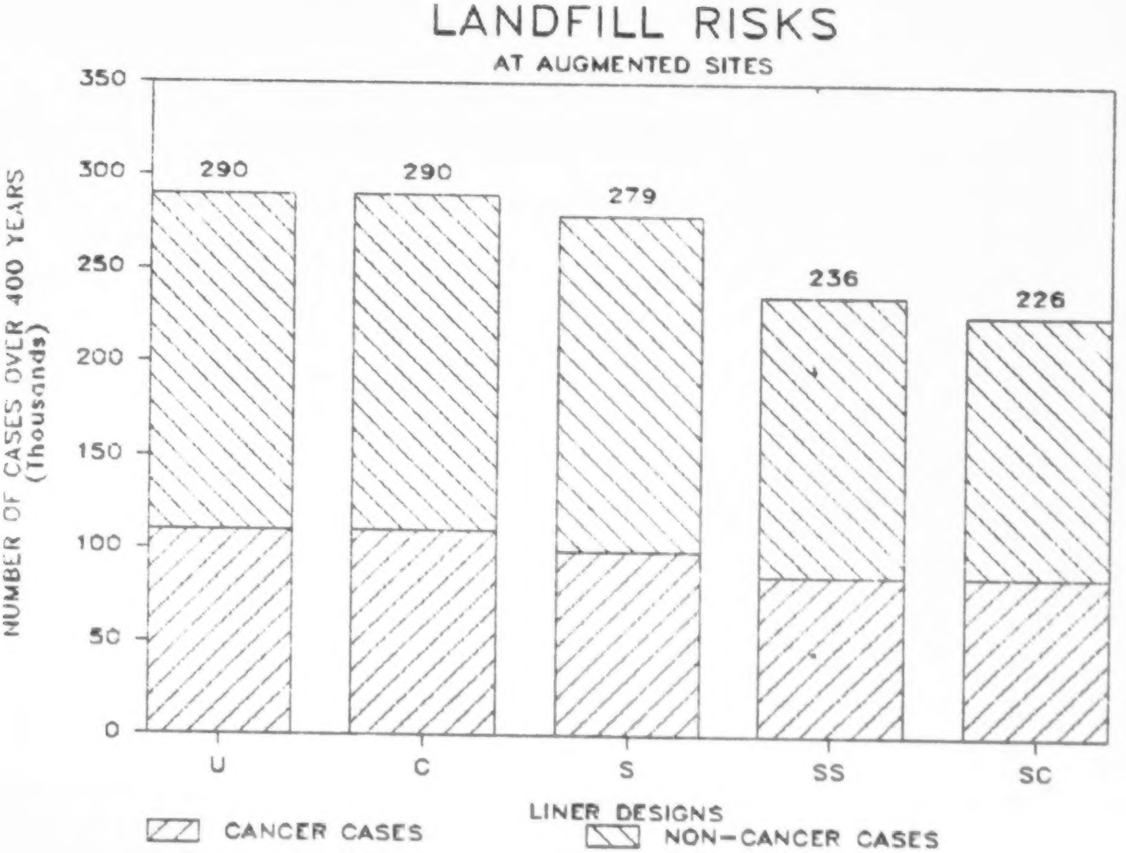
Our results for this application, as illustrated in Figure 4-2, showed that aggregate cases over 400 years at landfills were substantial, and that increasing the stringency of the liner requirements abated only a small portion of the risks. As the exhibit indicates, the majority of the cases are not cancer cases, and increasing the stringency of the requirements seems to be about equally effective (or ineffective) in reducing both cancer and noncancer cases. Utilization of a time period other than 400 years would produce different results.

Most of the cases occur at a small number of sites. Figure 4-3 illustrates this point; it shows cumulative cases with a single synthetic liner plotted against the number of facilities. (Plots of cumulative cases vs. number of facilities for other designs follow a similar pattern.) One facility alone accounts for 33 percent of the cases; the 46 most innocuous facilities account for less than one percent of the total.

Risk from surface impoundments follow a pattern similar to risks from landfills, as Figure 4-4 illustrates. Aggregate cases are high, and liners prevent only a portion of these risks. As with landfills, noncancer cases represent a majority of the cases.

Again, most of the risk is concentrated at a few sites. One site alone accounts for 46 percent of the cases for facilities with single synthetic liners; five sites account for 98 percent of the cases. The most innocuous 25 sites account for less than one percent of the cases. Figure 4-5 illustrates this point, showing the cumulative case distribution for surface impoundments with single synthetic liners. Again, the other designs display a similar pattern.

Figure 4-2



U = Unlined facility
C = Clay liner
S = Single synthetic liner
SS = Double synthetic liner
SC = Synthetic/clay liner

Figure 4-3

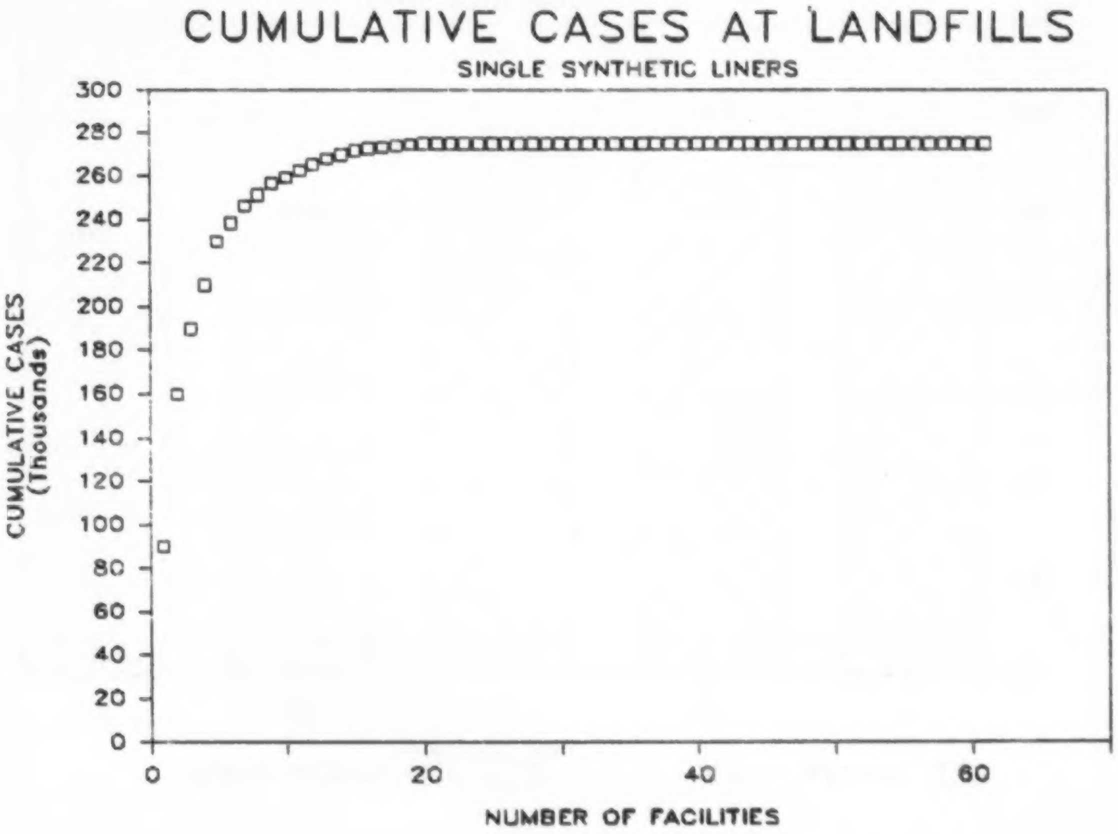
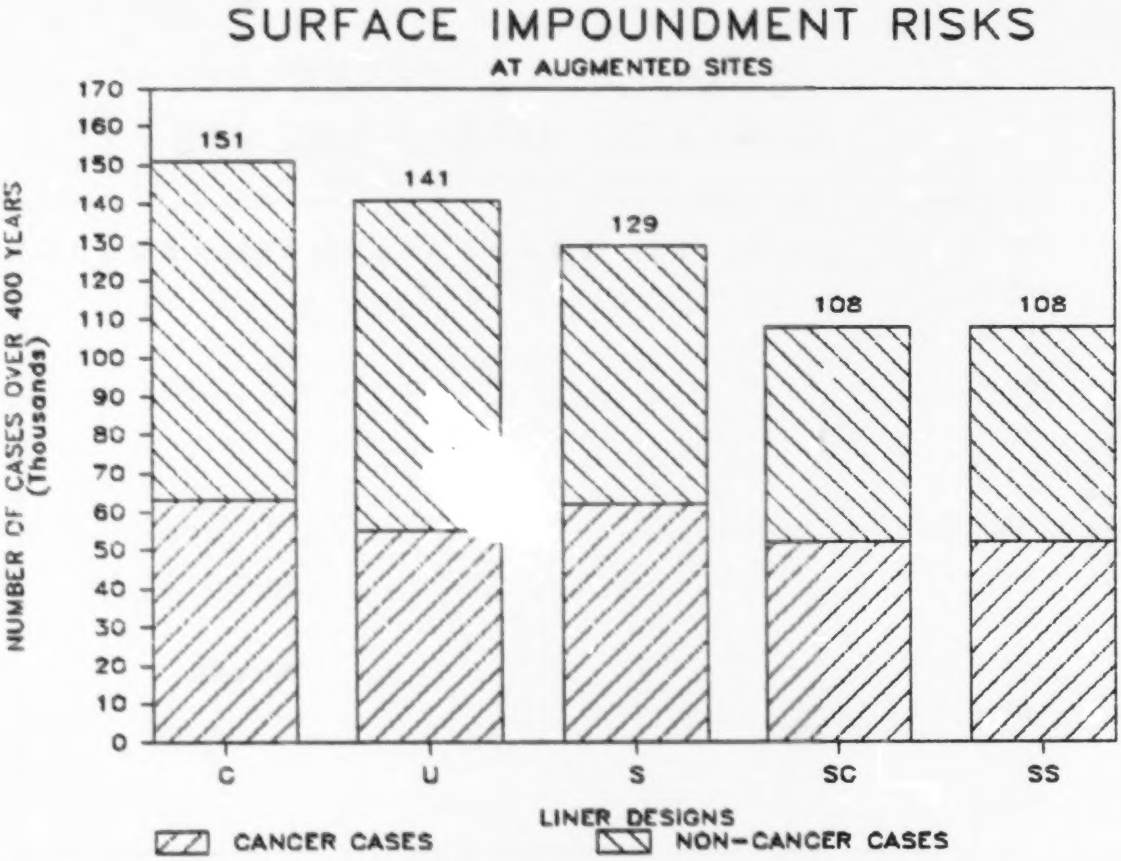
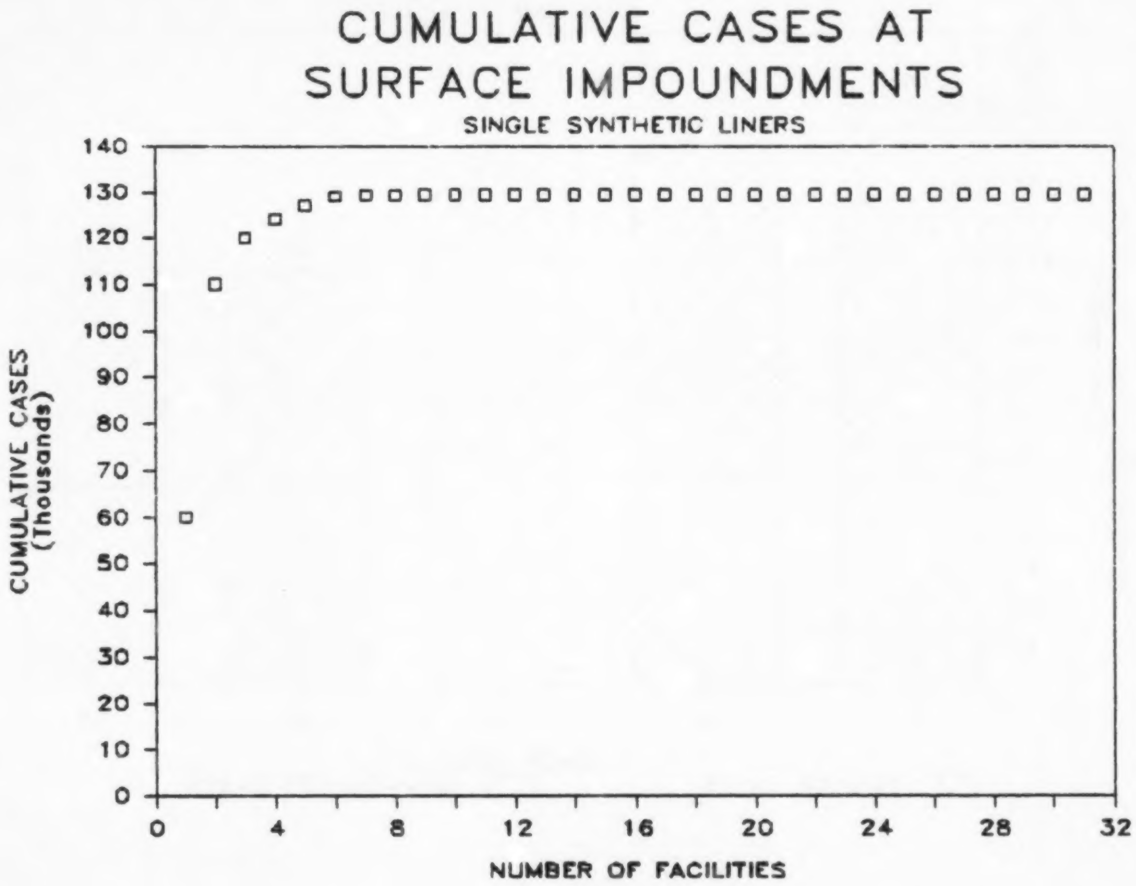

CUMULATIVE CASES AT LANDFILLS
SINGLE SYNTHETIC LINERS
CUMULATIVE CASES (Thousands)
300
280
260
240
220
200
180
160
140
120
100
80
60
40
20
0
0
20
40
60
NUMBER OF FACILITIES

Figure 4-4



C = Clay liner
U = Unlined facility
S = Single synthetic liner
SC = Synthetic/clay liner
SS = Double synthetic liner

Figure 4-5


CUMULATIVE CASES AT
SURFACE IMPOUNDMENTS
SINGLE SYNTHETIC LINERS
CUMULATIVE CASES
(Thousands)
140
130
120
110
100
90
80
70
60
50
40
30
20
10
0
0
4
8
12
16
20
24
28
32
NUMBER OF FACILITIES

By combining cost information for the disposal facilities with risk information, we can produce cost-effectiveness curves, and cost/risk ratios. Figure 4-6 shows the cost-effectiveness curve for landfills. Clay designs are not on the curve, since single synthetics are more protective and cost less. (In interpreting this result remember that we assumed that hazardous waste constituents would not adsorb to the liners. Had we assumed that they would, and that the wastes would not be incompatible with the liners, clay liners would have been more effective.) Double synthetic liners are not on the curve either as we have drawn it, since they are slightly more expensive and slightly less effective than synthetic/clay liners, but given the uncertainties accompanying the risk estimates, the difference in risk reduction is probably not meaningful.

From the costs and risks underlying the cost effectiveness curves, it is also possible to estimate cost/risk ratios. Moving from unlined to synthetic lined facilities costs about $35,000 per case avoided, while moving from single synthetic liners to synthetic/clay costs less than $1000 per case.

We also prepared a cost-effectiveness curve for surface impoundments, as Figure 4-7 shows. As with landfills, clay designs are not on the curve, nor are double synthetic liners, which are slightly more expensive than synthetic/clay liners but no more protective. Again, given the uncertainty in the risk modeling, the difference in cost is not meaningful.

We also calculated cost/risk ratios for surface impoundments. Moving from an unlined to a single synthetic design costs about $7,500 per case avoided, while moving from single synthetic to synthetic/clay costs less than $400 per case.

Although these cost/risk ratios are attractive, tailoring the regulations could produce even better ratios. For example, 43 percent of the unlined landfills and 31 percent of the unlined surface impoundments showed no risks. Exempting these facilities from liner requirements would reduce costs substantially while leaving risks unchanged.

Figure 4-6

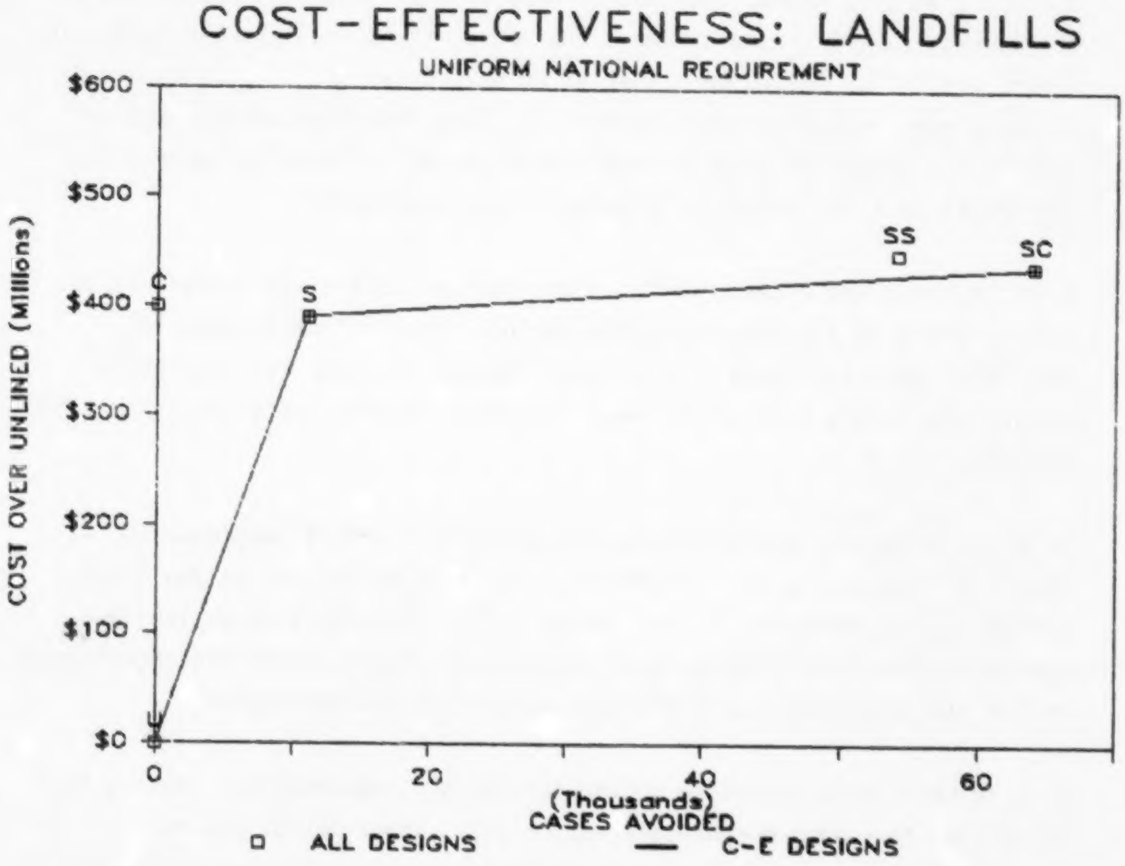
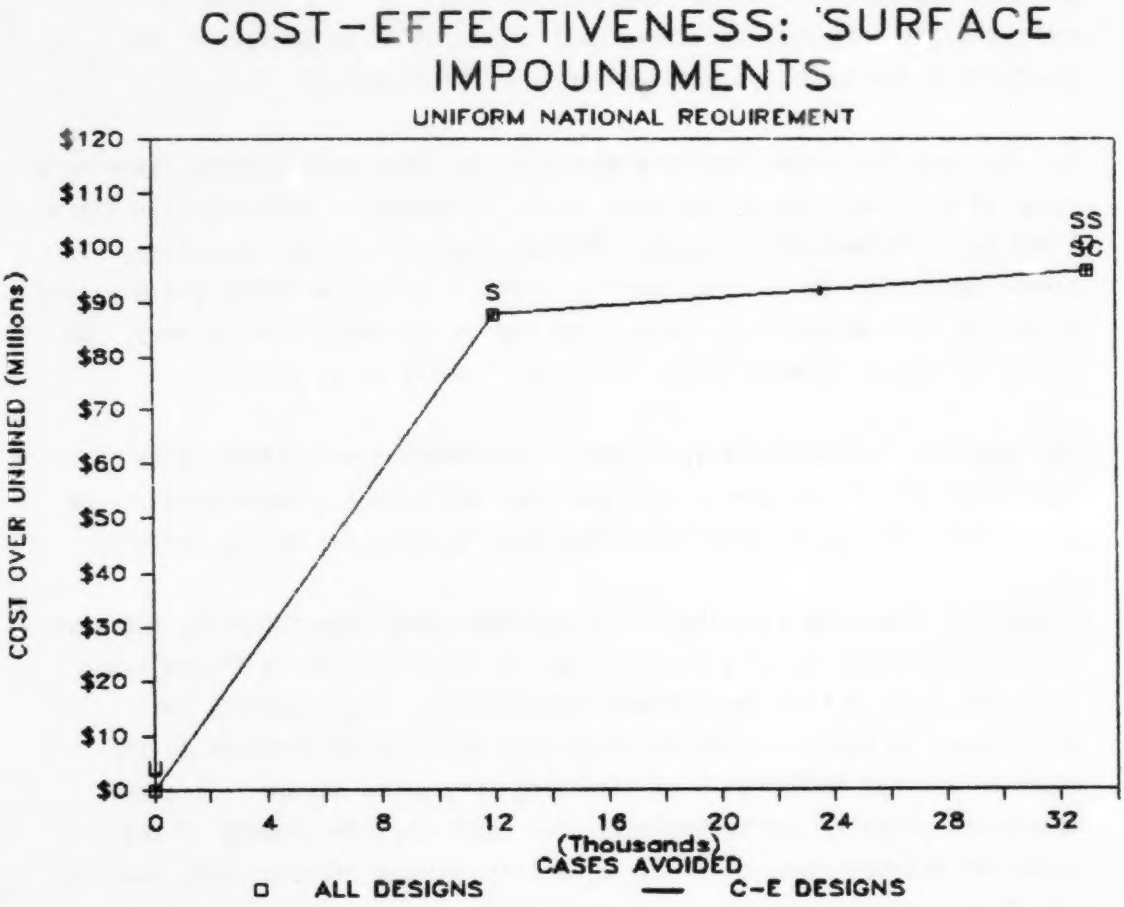

COST-EFFECTIVENESS: LANDFILLS
UNIFORM NATIONAL REQUIREMENT
$600
$500
$400
$300
$200
$100
$0
COST OVER UNLINED (Millions)
C
U
S
SS
SC
0
20
40
60
(Thousands)
CASES AVOIDED
□ ALL DESIGNS
— C-E DESIGNS

Figure 4-7


COST-EFFECTIVENESS: SURFACE IMPOUNDMENTS
UNIFORM NATIONAL REQUIREMENT
COST OVER UNLINED (Millions)
$120
$110
$100
$90
$80
$70
$60
$50
$40
$30
$20
$10
$0
0
4
8
12
16
20
24
28
32
(Thousands)
CASES AVOIDED
U
S
SS
SC
ALL DESIGNS
C-E DESIGNS

4.2 Comparing the Risk of Environmental Settings

In our second application we examined the effect that environmental setting has on risk. Holding technology constant (double synthetic liners), we examined the distribution of risk to exposed populations under varying climate, hydrogeologic setting, and well distance scenarios. We selected three waste constituents and examined risks to individuals under all 135 possible permutations of environmental setting. (Five climates, nine hydrogeologic settings, and three well distances.) We performed the simulations for landfills and for surface impoundments.

In selecting the wastes for this analysis, we chose constituents spanning a range of mobility classes and toxicities: chromium is from mobility class 1 and has a below-average potency for the chemicals in our data base; carbon tetrachloride is from mobility class 2 and has a higher potency; and parathion is from mobility class 3 and has an extremely high potency. We did so to obtain as much diversity in our results as possible.

Our hypothetical landfills and surface impoundments were large, with capacities of 123,000 metric tons per year and 5 acres, respectively. We assume that the waste constituent comprises ten percent of the capacity.

Figure 4-8 shows the distribution of average risks from chromium, the most mobile constituent in this application, in landfills, while Figure 4-9 shows the distribution for surface impoundments. They indicate that differences in environmental settings lead to large differences in the risks to exposed individuals of disposing of a waste stream in a given management practice and technology. For landfills, 39 percent of the scenarios produce risks of 10^{-1}, 36 percent produce risks of less than 10^{-1} and 26 percent produce risks less than 10^{-9}, i.e., no risk. As the figures show, one third of the scenarios for surface impoundments produce risks of 10^{-1} for exposed individuals, while slightly less than one third of the scenarios produce no risks.

Figures 4-10 and 4-11 show the comparable plots for carbon tetrachloride. Carbon tetrachloride is less mobile but more toxic than chromium; its distribution shows less risk. In both surface impoundments and landfills

Figure 4-8

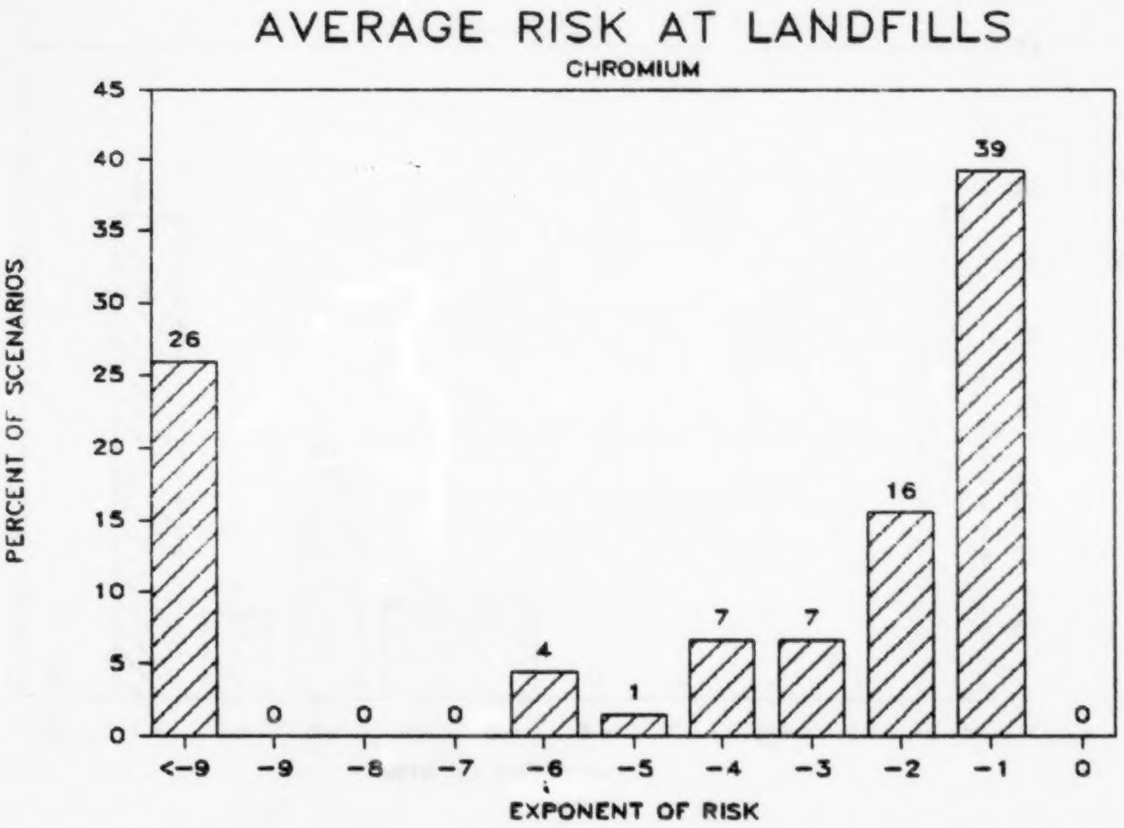

AVERAGE RISK AT LANDFILLS
CHROMIUM
PERCENT OF SCENARIOS
45
40
35
30
25
20
15
10
5
0
26
0
0
0
4
1
7
7
16
39
0
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
EXPONENT OF RISK

Figure 4-9

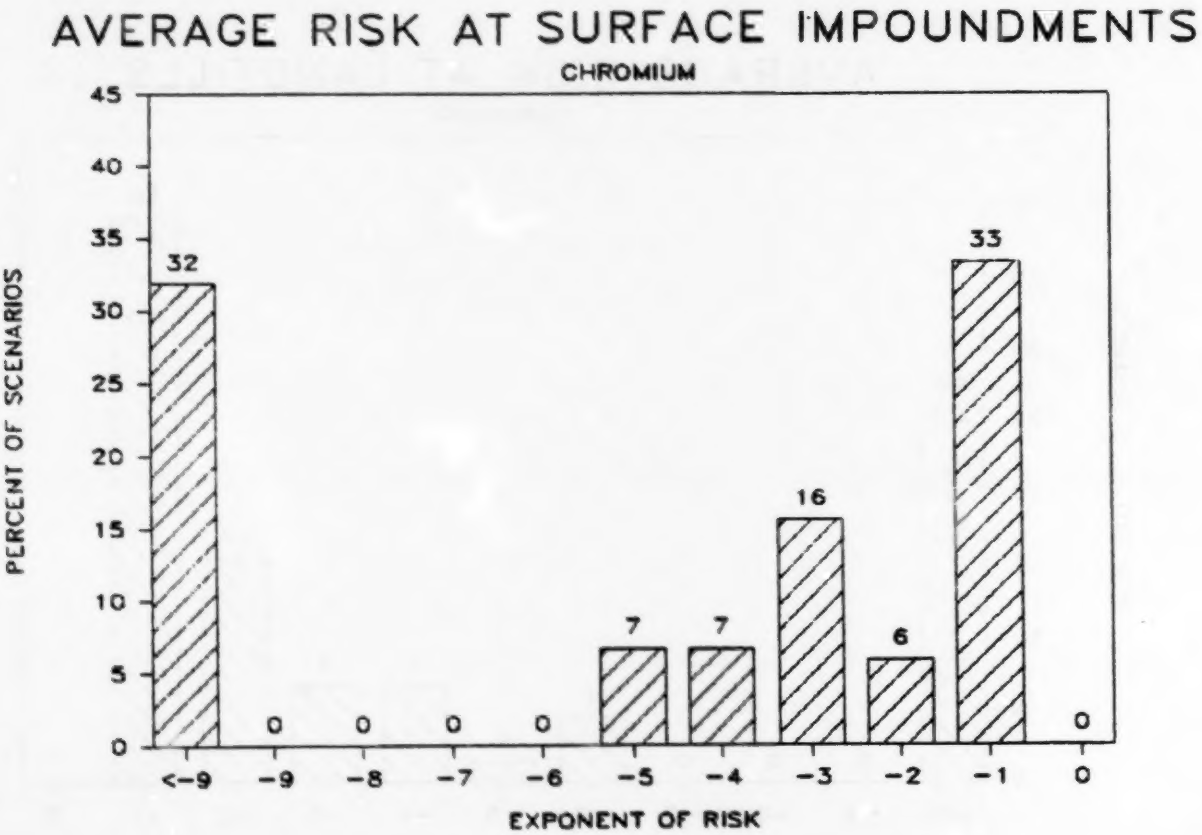

AVERAGE RISK AT SURFACE IMPOUNDMENTS
CHROMIUM
PERCENT OF SCENARIOS
45
40
35
30
25
20
15
10
5
0
32
0
0
0
0
7
7
16
6
33
0
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
EXPONENT OF RISK

Figure 4-10

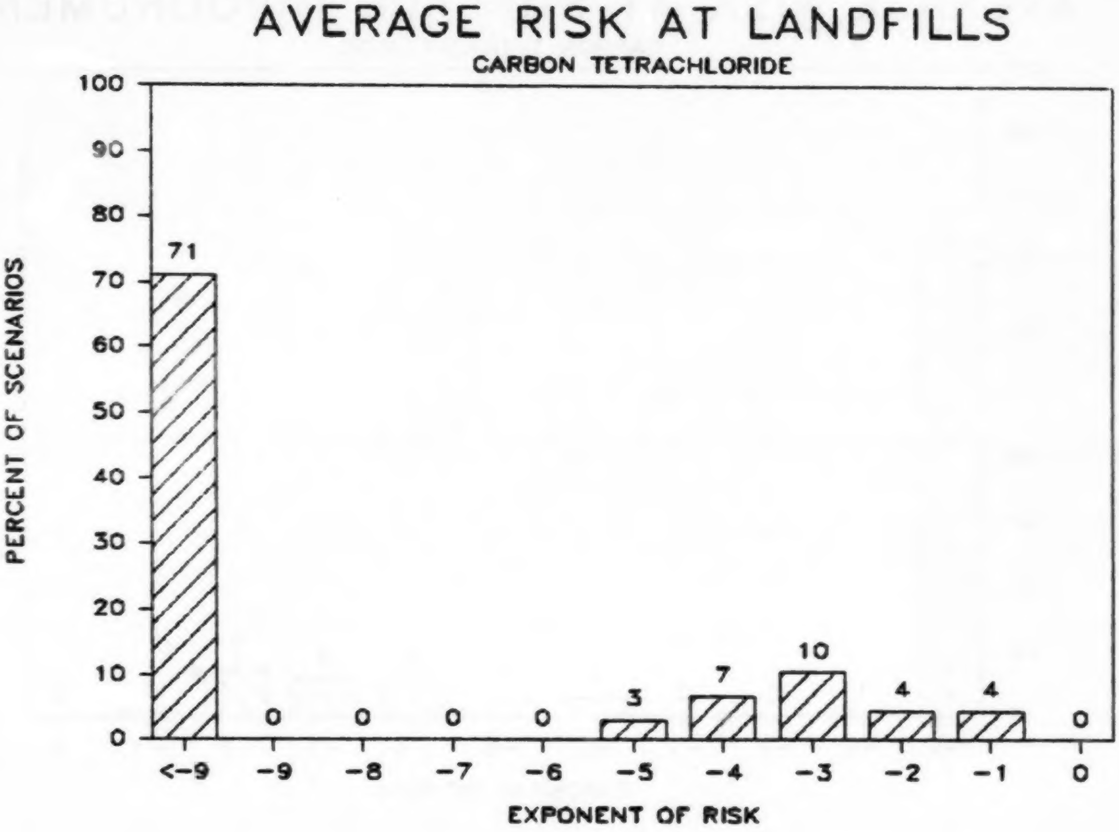
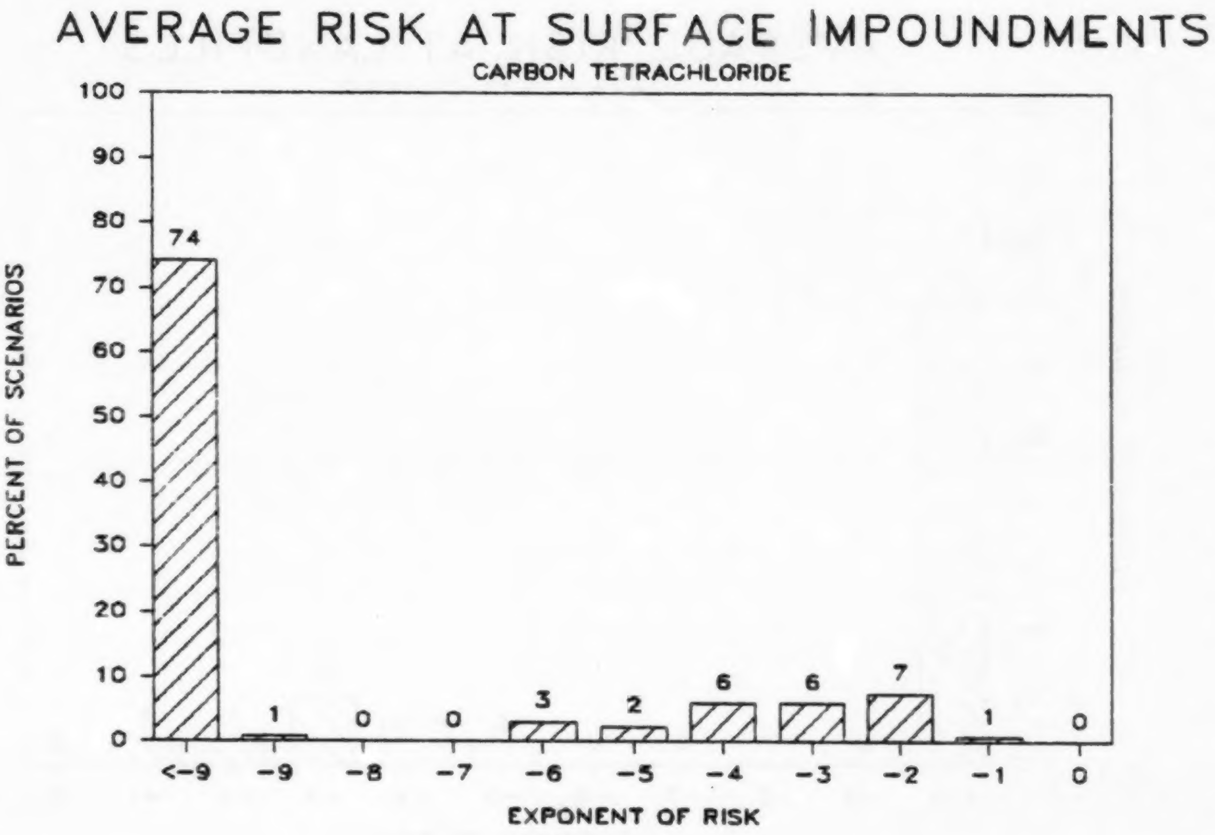

AVERAGE RISK AT LANDFILLS
CARBON TETRACHLORIDE
PERCENT OF SCENARIOS
100
90
80
70
60
50
40
30
20
10
0
71
0
0
0
0
3
7
10
4
4
0
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
EXPONENT OF RISK

Figure 4-11


AVERAGE RISK AT SURFACE IMPOUNDMENTS
CARBON TETRACHLORIDE
PERCENT OF SCENARIOS
100
90
80
70
60
50
40
30
20
10
0
74
1
0
0
3
2
6
6
7
1
0
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
EXPONENT OF RISK

almost three quarters of the scenarios produced negligible (10^{-9} or less) risk; most of the remaining scenarios, however, produced average risks of 10^{-5} or larger to exposed individuals.

We do not present the plots for parathion because none of the scenarios produced any exposures; due to its low mobility, parathion never broke through the unsaturated zone during our simulation period.

In addition to indicating that environmental setting causes large differences in risks spanning more than 10 orders of magnitude, this application also begins to suggest how the three environmental factors we varied are important in determining risk. The effect of climate is the most apparent.

Drier climates produce less risk; in dry climates there is less leachate and therefore releases from failures are smaller. In this application, the risk in the direct climate is zero for all three constituents in both technologies because facilities never fail in this climate. (Recall that failure is defined as the release of leachate.) In the next driest climate, some facilities fail, but only chromium, the most mobile constituent, produces any risk at all. Carbon tetrachloride releases take over 1,000 years to travel through the unsaturated zone in this climate and therefore do not produce risks within the 400 year timeframe used for this analysis.

The data also illustrate the effect of the well locational distance on risk as zero-case scenarios are far more likely as we increase the distance to wells from 60 meters to 1,500 meters.

Finally, if we control for everything except hydrogeology, we can observe what effect the aquifer type has on risk. As we move from aquifers in which transport is slow (e.g. Generic Flow Field A--one meter per year) to the faster ones (Field E--10,000 meters per year), we observe that the peak and average constituent concentrations decrease, the period of exposure is less, and the average risk declines anywhere from two to as much as five orders of magnitude.

The decrease in concentration, and hence risk, in faster aquifiers results from greater dilution that occurs once the release enters the aquifer. In the fast aquifers we modeled, the flow of ground water is several orders of magnitude greater than in the slower ones, so, once in the ground, the leachate is diluted in a much greater flow. Similarly, because the leachate moves more quickly through the faster flowing aquifers, the exposure both begins sooner and ends sooner, resulting in shorter periods of exposure. As a result of these two factors, the average risk declines substantially in faster aquifers.

If, however, groundwater flow is so slow that constituents, given their mobility, don't reach exposure points, given their distance, then slow flows will produce smaller risks.

In summary, this application shows that risks are extremely sensitive to environmental setting; when combined with the evaluation of national technology standards, it shows risks are far more sensitive to changes in environmental settings than to changes in technology.

4.3 Comparing the Risk of Different Wastes

To compare the risks of different wastes, we simulated risks for 15 waste streams in hypothetical landfills and 30 waste streams in hypothetical surface impoundments, holding liner technology and environmental setting constant, and assuming that people drink the contaminated groundwater.

We selected waste streams by examining responses to OSW's mail survey. We modeled the risk for a waste stream either in landfills, surface impoundments, or both if (1) it appeared in landfills, surface impoundments, waste piles, underground injection, or land treatment as one of any facility's largest ten waste streams, (2) it can be managed in landfills or surface impoundments, and (3) we could characterize its constituents and find the necessary physicochemical properties to simulate its risks. Figure 4-12 shows the EPA waste codes, the constituents of concern, and their concentrations for this application.

Figure 4-12

WASTE STREAMS USED IN LINER LOCATION RISK AND COST ANALYSIS MODEL

EPA Waste ID	Constituents of Concern	Concentration (ppm)
D007	Cr	500
D008	Pb	500
D009	Hg	20
F001	Trichloroethylene	150,000
	Tetrachloroethylene	120,000
	1,1,1-Trichloroethane	270,000
	Dichloromethane	59,000
F002	Tetrachloroethylene	280,000
	1,1,1-Trichloroethane	78,000
	Dichloromethane	210,000
	Chlorobenzene	5,500
F003	Methyl Isobutyl Ketone	10,000
	Ethylbenzene	10,000
F006	Cu	1,800
	Cr(VI)	680
	Cd	350
	Pb	400
	Ni	3,650
F007	Cd	100
	Cu	140
	Cyanide	75
	Cr(VI)	170
	Ni	1,500
K011	Acrylonitrile	400
	Cyanide	3,000
	Acetonitrile	6,000
K013	same as above	
K014	same as above	
K046	Pb	5,000
K048	Cr(VI)	8
	Pb	1

Figure 4-12 (continued)

EPA Waste ID	Constituents of Concern	Concentration (ppm)
K049	Cr(VI)	6
	Pb	3
K050	Cr(VI)	10
	Pb	28
K051	Cr(VI)	4
	Pb	75
K052	Cr(VI)	1
	Pb	630
K060	Cyanide	7,400
	Phenol	1,000
	As	1,100
K061	Cd	890
	Cr(VI)	570
	Pb	13,500
K062	Cr(VI)	10
	Pb	100
P030	Cyanide salts	500,000
P070	Aldicarb	100,000
U002	Acetone	990,000
U044	Chloroform	980,000
U122	Formaldehyde	500,000
U127	Hexachlorobenzene	200,000
U128	Hexachlorobutadiene	900,000
U130	Hexachlorocyclopentadiene	900,000
U188	Phenol	850,000
U210	Tetrachloroethylene	999,000
U211	Tetrachloromethane	995,000

We simulated risk in a hypothetical landfill (for solids and sludges) with a capacity of 123,000 metric tons per year, 70 percent of which is waste. For surface impoundments, we selected a five acre impoundment capable of receiving 20,000 metric tons per year of liquids and sludge wastes. We configured both our hypothetical landfill and our hypothetical surface impoundment to have a single synthetic liner. We "disposed" of only a single waste stream in each facility.

We selected a moderately wet climate (10 inches of net infiltration per year) as the site of our hypothetical facilities, and located them over a fairly fast flowing aquifer (100 meters per year). The unsaturated zone is 2.5 meters for landfills and 5 meters for surface impoundments. We assumed that people drink from a well 60 meters down gradient of the site.

Our results from this application showed there is wide variability in risk for different waste streams holding all other factors constant. In landfills (Figure 4-13), eight of 15 wastes we modeled showed average risks of less than 10^{-9}, yet all but one of the remaining seven waste streams showed risks of 10^{-4} or greater. In surface impoundments (Figure 4-14), nine of 30 waste streams modeled showed risks of less than 10^{-9}, while 17 showed risks of 10^{-4} or more.

These results, combined with those from the evaluation of national technology standards, seem to indicate risk is far more sensitive to changes in waste stream composition than it is to changes in technology.

4.4 Implications

Our applications, while not definitive, illustrate that human health risks from land disposal vary over many orders of magnitude, principally as a function of environment and waste stream. There appear to be subtle and complex interrelationships among the many factors which comprise wastes, environments, and technologies. Thus, regulations appropriately tailored to these interrelationships could protect human health better and cost less than regulations based solely, or independently, on technology, waste, or location.

Figure 4-13

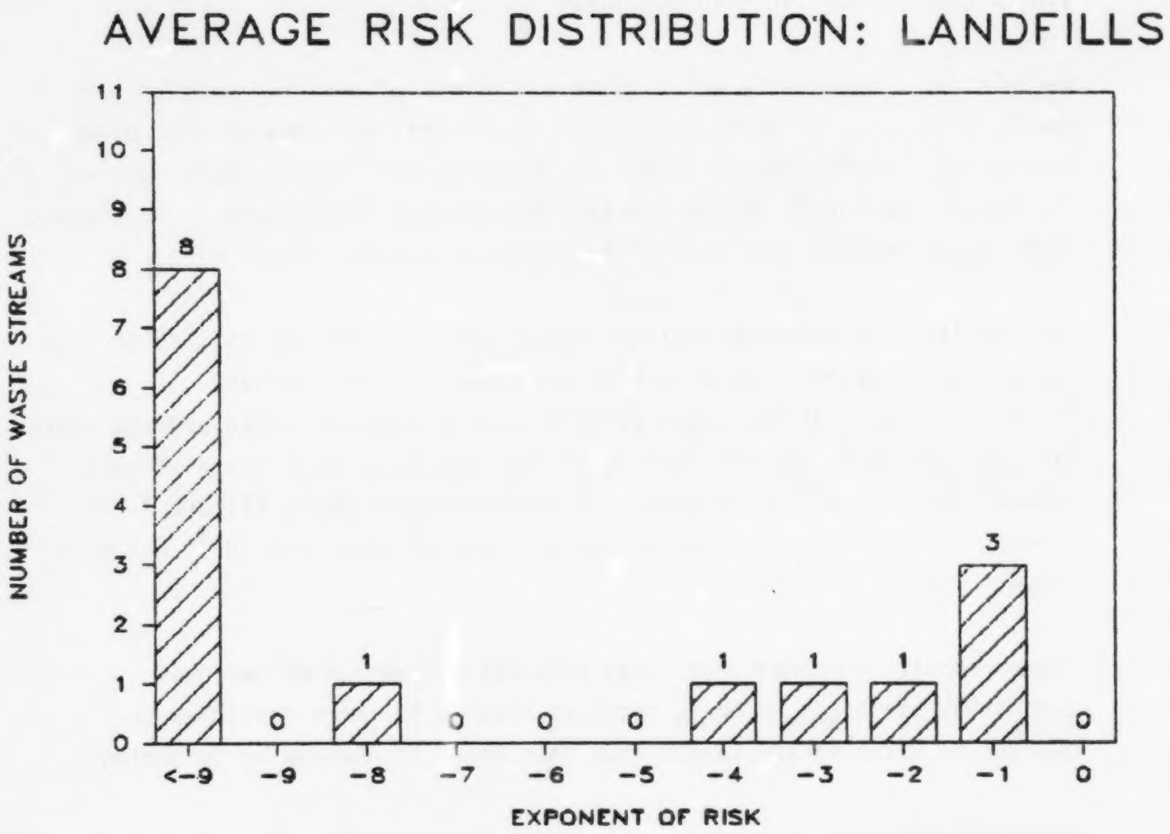
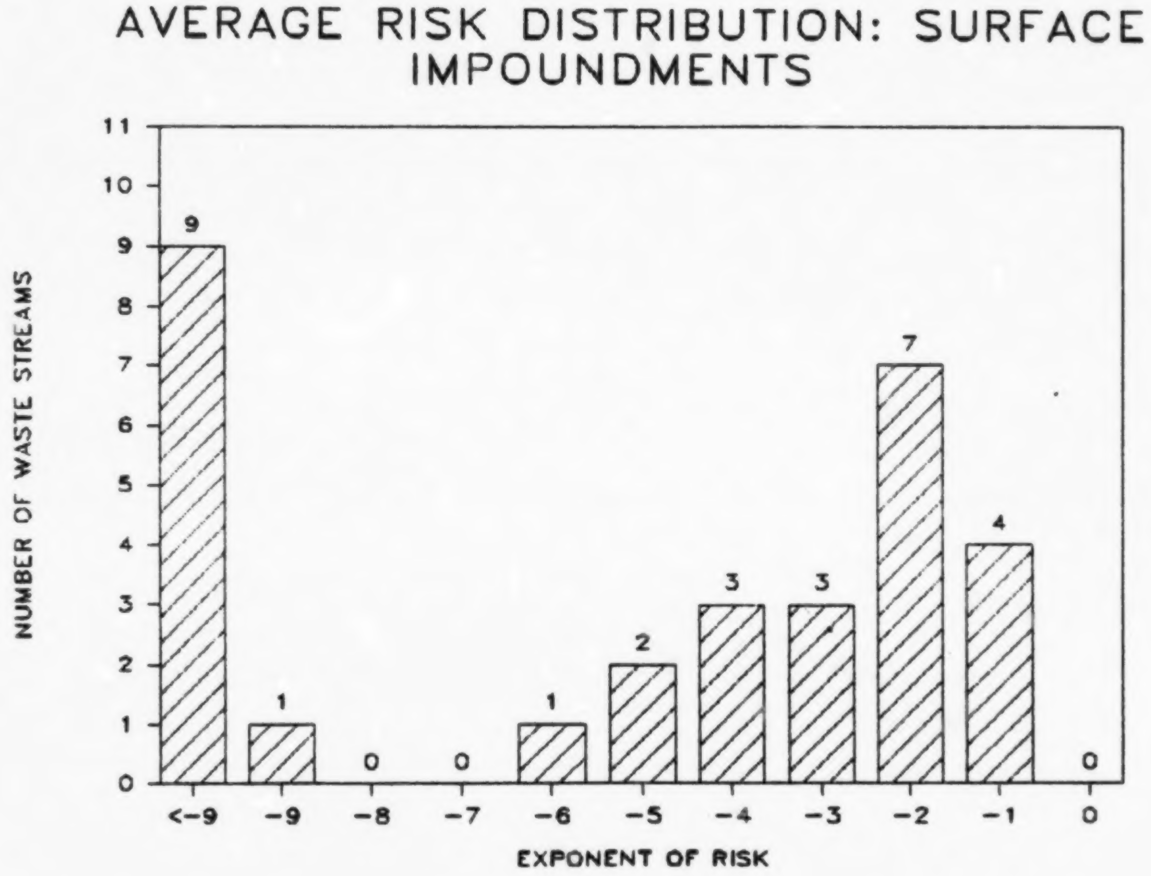

AVERAGE RISK DISTRIBUTION: LANDFILLS
NUMBER OF WASTE STREAMS
EXPONENT OF RISK
0
1
2
3
4
5
6
7
8
9
10
11
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
8
0
1
0
0
0
1
1
1
3
0

Figure 4-14



AVERAGE RISK DISTRIBUTION: SURFACE IMPOUNDMENTS
NUMBER OF WASTE STREAMS
0
1
2
3
4
5
6
7
8
9
10
11
9
1
0
0
1
2
3
3
7
4
0
<-9
-9
-8
-7
-6
-5
-4
-3
-2
-1
0
EXPONENT OF RISK

5. PLANNED MODEL IMPROVEMENTS AND APPLICATIONS

We have not currently completed sensitivity analyses, and have not finished examining all assumptions and limits to determine the constraints they impose on using the model for policy purposes. In this chapter we report on the next steps in the project: improving the model, validating it, performing additional applications, and preparing a second report on these tasks.

5.1 Improving the Model

The project team is currently making improvements in the model. In building and applying the model we have noticed several areas where we can improve it; in some cases we are now planning the improvements and in other cases we are already modifying the model to incorporate them. Additional sensitivity analyses will probably identify other areas with potential for improvement.

5.1.1 Improving the Failure and Release Submodel

Currently, planned improvements to the failure and release model fall into three areas: examining the underlying distributions, improving the hydrologic budget components, and modifying the basic structure of the model itself.

We will perform sensitivity analyses to determine how important the underlying probability distributions are in determining failures and releases. We will re-examine those distributions that are important, to assure that we have modeled them correctly.

In the current version of the failure and release submodel, the leachate collection system (as long as it functions) removes leachate from the facility, but we assume that it reinjects it into the facility. We are currently modifying the program to provide an option for removing the leachate. Thus, under this option, the total mass available to leach or volatilize from the facility will be the total mass present in the facility less the quantity collected by the leachate collection system. We will also investigate the effects on releases and risk of repairing and replacing failed cover components during the post-closure period.

Finally, we will modify the basic structure of the submodel itself. In the present version of the model, we calculate the percentage of the facilities that have failed at seven benchmark times and the average volume of leachate released given that failure occurs, using a Monte Carlo simulation and parsing the fault tree 1,000 times. In the refined versions, we will report statistics associated with these estimates so that we will be able, for example, to distinguish between catastrophic and noncatastrophic failures.

5.1.2 Waste Mass Inputs to the Unsaturated Zone

We will improve the assumptions on waste mass throughput in disposal facilities. We currently calculate mass input to the unsaturated zone by multiplying the volume of water passing through the facility by the aqueous solubility of the waste constituent. We do not account for degradation of constituents in the facility before release to the unsaturated zone. We are currently working on ways to model degradation in the time before releases begin.

In addition to this improvement, we are trying to identify ways to model more realistically waste releases in surface impoundments. We currently assume that all mass deposited in the impoundment is available to either volatilize or leach through to the unsaturated zone. This is unrealistic because most impoundments have surface water discharges; if the waste constituent can leach at the solubility limit, then the surface water discharge will remove some of the wastes. Moreover, after an owner closes an impoundment and solidifies the waste, it is not realistic to expect that the leachate will have concentrations at the solubility limits. We are planning to evaluate data on input/output relationships for surface impoundments to determine whether we can reasonably generalize on the mass balance of wastes in these systems. We are also planning to review information on the characteristics of impounded wastes after solidification to determine whether we can improve our assumptions in this area.

5.1.3. Improving the Subsurface Transport Submodel

We are considering several improvements in the subsurface transport submodel. The unsaturated zone model component only accounts for time delay; in our next version, we will also consider other processes that reduce contaminant concentrations or toxicities. The subsurface transport component is two-dimensional and therefore is likely to overestimate concentrations because it does not allow for dilution in the third dimension. We will investigate ways of modifying this component to obtain more realistic estimates of groundwater contaminant concentrations.

In the next phase of the project, we will model dilution at the well. Currently, we assume that the concentration of contaminant in the well will be identical to the concentration of contaminant in the ground water, ignoring the fact that most wells draw from different depths in the aquifer (not all of which will be contaminated) and that their cone of influence is likely to contain some uncontaminated water.

The current subsurface transport submodel describes flows in homogeneous, isotropic aquifers. Most aquifers, however, are not homogeneous and isotropic. Those that are not may have higher or lower concentrations of contaminants, and hence higher or lower risk. In particular, we will investigate how to apply the submodel to simulate flows in fractured media.

5.1.4 Improving the Exposure Submodel

In the current version of the submodel we make a worst case assumption that critically influences our risk results: we assume that people drink only contaminated water regardless of taste or odor thresholds. If we believe that people will not knowingly do so, then we must implicitly assume that they do not know the water is contaminated. Thus, we are implicitly assuming that groundwater monitoring does not work and that contaminants are present in drinking water at concentrations below their odor, taste, and physical appearance thresholds.

We have not yet resolved the issue of the effectiveness of groundwater monitoring at the facility as required by Subpart F of the Part 264 and 265 regulations, or of corrective action. We could model its effectiveness stochastically, and build this component into the submodel. For example, if we model it as effective fifty percent of the time, then in half of the cases where there is groundwater contamination, monitoring will detect it and consequently people will not drink the water.

We will also determine the odor, taste, and physical appearance thresholds for the contaminants we model, and build logic into the program to calculate separately the risks from concentrations above and below these thresholds. We will also attempt to collect better information on populations using ground water for our site visit sites and for the mail survey sites.

Finally, we will add a corrective action component to the model. Thus, if contamination is detected through monitoring, odor, appearance or taste, we will model the effects of clean-up on risk and on cost.

5.1.5 Improving the Hazard Submodel

Our next steps to improve the hazard submodel will cover four areas: (1) verifying some of the basic assumptions underlying the risk estimation algorithm for noncarcinogens, (2) comparing our risk estimates with estimates derived from other systems, (3) improving the components of the algorithm, and (4) determining the sensitivity of the results to changes in some of the input values. These tasks are described briefly below.

Two potentially controversial assumptions underlie the dose-response model: that the dose-response function is not linear for noncarcinogens (i.e., k = 2 for most substances, and k = 4 for several others), and that graded responses can be converted to quantal responses by use of a scaling factor. More work is needed to corroborate the initial analysis that derived an empirical value for k. We also need to review our use of scaling factors for all noncarcinogenic effects, particularly whether it is appropriate to base a scaling factor on the assumption that, for graded effects, the entire test population responds at the minimum effective dose.

We intend to compare risk estimates using our risk estimation algorithm with estimates derived using other systems, such as the one EPA's Integrated Environmental Management Division (IEMD) has developed. For noncarcinogens, IEMD uses dose-response functions that show zero response at doses below a threshold, and linear dose-response (up to 100 percent response) at doses above the threshold. We will also compare our results to the results obtained by using the more traditional acceptable daily intakes (ADIs).

We believe that we can improve several components of the risk algorithm. Recent work on the influence of the duration of exposure on the dose-response relationship may be applicable to the model. This work indicates that there may be different risks for the same effect and same average lifetime exposures, depending on whether these exposures are comprised by high doses for short periods or low doses for longer periods. We may also adjust our risk estimates by including a judgmental severity factor to account for differences in the harshness of the different health effects. We will investigate existing severity scales to see whether they can be applied to the Model.

Finally, we will perform some sensitivity analyses on several key parameters. We may test the sensitivity of the risk results to differences in our values for "k", the exponent of dose in the dose-response function. We may also test the importance of using maximum likelihood estimates of carcinogenic potency rather than the upper 95 percent confidence limits on potency. Other candidates for sensitivity analysis include our assumptions on interspecies extrapolation (e.g., scaling by surface area rather than by body weight), time-dependent exposures, and scaling factors.

5.1.6 Improving the Cost Model

The present version of the cost model does not include the expense of treating leachate from the facility. In our next version of the cost model, we will use the failure and release submodel to estimate the quantity of leachate from the leachate collection system. We will then estimate the capital and O&M costs for building and running a leachate treatment facility of the appropriate size, and include these costs in our estimates. If, for some facilities, it is less expensive to send the collected leachate off-site to a commercial treatment facility, we will include the cost of transport and commercial treatment of leachate in the costs of these facilities.

Since we will improve our failure submodel by considering the effect of repairing or replacing cover during the post closure period, we will also include the costs for these activities in the cost model.

5.1.7 Improving the Cost/Risk Modeling

To improve the cost/risk modeling, we will consider building in a logic component enabling us to discount risks as well as costs. This will let us test the sensitivity of the risk and the cost/risk results to the particular discount rate selected. We will also perform sensitivity analyses of the effects of the cost discount rate on costs and cost/risk ratios. Finally, we will examine the effects of calculating risks and cost/risk ratios over both shorter and longer time periods than the current 400 years. We expect that modeling for a 400 year time period blurs any advantages that enhanced technology may possess in shorter time periods. Over such a long time period almost all designs fail in all climates. On the other hand, modeling over this time period will result in higher risk for all those scenarios in which there will be increased exposure over the time periods.

5.2 Additional Applications

After improving the model, OSW will use it for additional policy analysis. RCRA reauthorization requires EPA to promulgate regulations mandating more stringent liner technology than the current regulations require. We will use the improved model in performing the RIA for these regulations. We also expect the model will be useful in policy analyses for location standards, for land disposal restrictions, for analysis of Subtitle D facilities, for underground tanks and many other uses.

5.3 Validating the Model

The Liner Location Risk and Cost Analysis Model depends on a large number of assumptions and simplifications. Accordingly, it is not possible to verify directly the performance of the model, at least for several of the principal components, because of the difficulty in measuring some of the complex processes we model (i.e., liner system failure, subsurface transport, and effects of environmental exposures on human health). However, we can infer the accuracy of several of the key elements by reviewing some types of empirical data. Some of the model validation activities we are planning include:

- Reviewing data on groundwater monitoring at permitted facilities to determine how often it detects leakage, and what types of pollutants it finds. This will help us to check the failure analysis and subsurface transport components of the risk model, as well as our assumptions on waste characterization.

- Reviewing groundwater concentration data at CERCLA sites. This will help verify critical assumptions about retardation factors, especially for inorganics. Our model predicts very long periods before low-mobility constituents reach wells even 60 meters from a source; if these constituents are present in plumes, we can calculate actual retardation factors by examining hydrogeologic data.

- Reviewing results of ongoing air monitoring at landfills and surface impoundments. We need to verify that our approximations of air emissions are realistic.

- Reviewing leachate monitoring data at facilities with leachate collection systems. This will allow us to check our assumption that wastes leach out of facilities at their solubility limits. It will also allow us to evaluate the collection efficiencies.

We also intend to analyze the sensitivity of the model to different parameters, determine which parameters are most important, systematically identify ways to check the realism of the model, and set priorities for validating the model according to the sensitivity results.

5.4 Phase II Report

When we complete the work described above, EPA will issue a second report. This report will explain the model modifications that incorporate the improvements we discussed previously and that respond to comments from reviewers.